HEADWATER STREAM INVERTEBRATE COMMUNITIES:

A COMPARISON ACROSS ECOREGIONS AND LOGGING HISTORIES

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A

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General Abstract

Monitoring stream condition is not always conducted with understanding how climate may influence anthropogenic disturbances. Stream monitoring has traditionally been accomplished through sampling benthic invertebrates, while sampling drifting invertebrates as a potential monitoring tool has received little attention, in spite of drift often being easier and less expensive to sample. The objectives of this study were to understand how logging influences headwater stream invertebrate communities (benthic and drift) across two ecoregions in the Cascade Range, central Washington, and to determine whether drift samples might serve as a replacement for benthic samples in assessing headwater stream condition. Benthic and drifting invertebrates were sampled from 24 headwater streams in logged and unlogged watersheds within two ecoregions (wet and dry), and community metrics contrasted. Invertebrate community responses to logging varied with ecoregion (e.g., higher shredder densities in logged watersheds of wet ecoregion only). Differences in benthic community structure were not reflected in the drift, and relationships between benthos and drift were highly variable. Although both sampling types (benthic, drift) revealed ecoregional and land-use (logging) differences in invertebrate communities, lack of consistent relationships between the sampling types suggests drift sampling does not provide more reliable information about stream benthos or headwater stream condition.

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Acknowledgements

I dedicate this thesis to my father Robert James Medhurst who continually inspired me with his excitement and interest in science. He would remind me that the rough times "tested my mettle" and were an opportunity to see what I was really made of. I will forever cherish the day I called to tell him I had successfully defended shortly before his death. I will continue to live my life abiding by his life philosophy, "If you aren't going to do something right, don't do it at all".

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General Introduction

Headwater streams (first and second order) are abundant across many landscapes, representing as much as 80% of stream catchment area (Sidle *et al.*, 2000; Meyer & Wallace, 2001). These streams act as conduits for water, particulate organic matter and dissolved nutrients to larger order streams and downstream habitats (Vannote *et al.*, 1980; Meyer & Wallace, 2001). Individual streams, while abundant, have relatively small catchment areas and are easily influenced by small-scale differences in local conditions (Meyer *et al.*, 2007).

The variety of habitats found within headwater streams support an array of plant and animal life making headwater streams an important source of biodiversity (Meyer *et al.*, 2007). Aquatic invertebrates comprise a large proportion of the animal diversity in these systems and their movement among aquatic and terrestrial habitats can influence ecosystem processes and food web dynamics (Nakano *et al.*, 1999; Laeser *et al.*, 2005). It has been suggested that the downstream movement of invertebrates, via fluvial transport (invertebrate drift), may be an important means of food delivery for downstream consumers (Wipfli *et al.*, 2007). In addition, larval invertebrates play an important ecological role in the processing of organic matter in streams (Stout *et al.*, 1993). The reduction of coarse particulate organic matter to finer particles by shredding taxa increases breakdown rates of this material and provides resources for gathering and filtering taxa further downstream (Mulholland *et al.*, 1985).

The use of aquatic organisms in the biological assessment of stream condition has a long history (Kolkwitz & Marsson, 1908; Hilsenhoff, 1977, 1998), and the relative

abundance of taxa, or taxonomic groups, within aquatic communities has often been used to assess the level of disturbance or impacts to streams (U.S. EPA, 2004). Woodcock and Huryn (2007) suggested that the transport of energy to adjacent habitats may be impaired with increasing stress from disturbance. Forestry activities can influence the quality and quantity of riparian vegetation fed upon by in-stream primary consumers such as shredders. These activities can also alter the amount of solar radiation reaching the stream surface and thus, alter the growth of benthic periphyton relied upon by algivorous taxa.

Secondary benthic production can also be a useful measure of relative stream condition and is rarely measured directly, but rather, inferred from instantaneous samples of stream biota (Woodcock & Huryn, 2007). Traditionally, benthic invertebrates have been used in bioassessment because the sampling techniques and data analysis are well developed (Resh *et al.*, 1996). However, some researchers have suggested that drifting invertebrates may be a useful tool in estimating secondary benthic production (Siler *et al.*, 2001), and may hold promise for assessing the degree of disturbance or stream condition.

The purpose of this study was to (1) determine if benthic invertebrate communities in headwater streams differ according to ecoregion and logging history, (2) establish whether headwater invertebrate drift assemblages reflect associated benthic communities, and (3) determine whether drift samples taken from headwater streams can be used to monitor headwater stream condition. Although this study was conducted in the Wenatchee River Subbasin of the eastern Cascade Range, Washington, results are

expected to be relevant to headwater systems in other mountainous regions, and to larger stream systems as well.

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CHAPTER 1

Benthic invertebrate community structure across ecoregions and logging histories

1.1 Abstract

1. Monitoring stream condition is not always conducted with an understanding of how climate may influence anthropogenic disturbance to these systems.

2. Headwater streams are tightly linked to riparian forests, and logging activities may influence the relative abundance of aquatic invertebrates which serve as a food resource to downstream and terrestrial riparian consumers.

3. The objective of this study was to understand how logging influences aquatic invertebrate populations across different ecoregions in the Cascade Range, central Washington.

4. Benthic invertebrates were quantitatively sampled from 24 small, headwater streams in logged and unlogged watersheds within two distinct (wet and dry) ecoregions, and analyzed for differences in ten community metrics.

5. Differences in community metrics between logged and unlogged watersheds were not consistently detected in both wet and dry ecoregions, likely due to differences in deciduous riparian forest canopies from different climates and post-logging forest succession.

6. These results suggest that invertebrate community response to logging varies by ecoregion. Understanding how these communities naturally vary with climate and landscape would improve monitoring efforts aimed at identifying differences resulting from anthropogenic disturbance.

Medhurst, R.B. (2007) Benthic invertebrate community structure across ecoregions and logging histories. Prepared for submission to *Freshwater Biology*.

1.2 Introduction

Natural disturbance plays a major role in the composition, functioning, and overall maintenance of aquatic ecosystems (Townsend *et al.*, 1997; Harding *et al.*, 1998; Lytle, 2001; Lenz *et al.*, 2004). It can alter refugia and food resource availability thus affecting the community composition of organisms relying on these resources (Lugthart & Wallace, 1992; Wallace *et al.*, 1999; Hernandez *et al.*, 2005). Anthropogenic disturbance, such as timber harvest, can also influence the structure and function of aquatic ecosystems by altering both physical and biological characteristics of streams and rivers draining forested ecosystems (Murphy *et al.*, 1986; Stout *et al.*, 1993; Swank *et al.*, 2001). Top down geo-climatic factors may influence the direction of post disturbance forest succession and play a role in the resulting aquatic community composition (Frissell *et al.*, 1986; Power, 1992).

Headwater streams (first and second order) are an important component of riverine systems, making up 70-80% of catchment areas (Sidle *et al.*, 2000; Meyer & Wallace, 2001) and accounting for up to 53% of linear riverine miles (Nadeau & Rains, 2007). Because of their large surface area to volume ratio and relatively narrow width (often < 1 m) they are tightly linked to riparian vegetation and the surrounding upland landscape. Headwater streams act as a conduit for the transfer of materials to downstream habitats (Vannote *et al.*, 1980; Meyer & Wallace, 2001) and potentially to consumers within those habitats (Piccolo & Wipfli, 2002; Wipfli & Gregovich, 2002).

The Wenatchee River subbasin has been clear-cut and selectively logged by both commercial and private landholders over the past 150 years (R. Simon, personal

communication, June, 2005). Forestry activities can influence the quality and quantity of riparian vegetation fed upon by in-stream primary consumers such as shredders. These activities can also alter the amount of solar radiation reaching the stream surface and thus, alter the growth of benthic periphyton relied upon by algivorous taxa. Removal of riparian forest and the resulting reduction in leaf litter inputs can cause a decline in shredder densities (Stout *et al.*, 1993; Stone & Wallace, 1998). Following timber harvest, increased sunlight in affected reaches can bolsters algal production (Murphy *et al.*, 1986). Stone and Wallace (1998) and Kiffney *et al.* (2003) found that higher algal production following timber harvest increased the abundance of invertebrate grazers. In addition, Stone & Wallace (1998) showed a subsequent reduction in grazers and a return of shredders with riparian regrowth and forest succession.

The taxonomic richness of headwater streams can number in the hundreds, making them an important source of biodiversity (Meyer *et al.*, 2007). Disturbance can influence this richness as proposed by the intermediate disturbance hypothesis (Connell, 1978), which states that aquatic invertebrate taxa richness is highest at intermediate frequencies or intensities of disturbance. With disturbance, new pioneer species can gain a foothold coexisting among existing taxa, reduced in number but not yet driven from the system. However, if the frequency or intensity of disturbance is too high, decreased taxonomic richness can occur (Death & Winterbourn, 1995) with a simultaneous increase in numeric density of the remaining tolerant taxa (Minshall *et al.*, 2001; Reed, 2003).

Environmental context should be considered when evaluating forestry impacts, which may otherwise be masked by natural variation among sites (Martel *et al.*, 2007). As

climate can constrain broad patterns of disturbance, macroclimate must be considered in regional comparisons (Hessburg *et al.*, 2000). In the case of forestry, post-logging forest succession is under strong climatic control, with available precipitation, solar radiation, and mean annual temperature contributing to the trajectory of that succession (Hessburg *et al.*, 1999).

River drainages in the Columbia River basin are influenced by Pacific maritime weather patterns that extend across the Cascade Mountains and by continental air masses in interior regions east of the Cascade Range (Wissmar, 2004). Vegetation cover, including upland forests drained by headwater streams, is largely governed by ecoregion characteristics such as mean precipitation, temperature, solar radiation, and underlying geology (Hessburg *et al.*, 2000). Rain-shadow effects near the crest of the Cascades can be pronounced in upland forested areas (Kovalchik, 1992) and are largely responsible for defining these ecoregions. Further, Hessburg *et al.* (2004) showed that ecological subregions of the eastern Washington Cascade Mountain Range readily explained the partitioning of subwatersheds by area in to historical fire severity classes and suggested that upland forest communities may differ as a result of long-term differences in regional fire frequency and intensity.

The objectives of this study were to determine if aquatic invertebrate communities differed between logged and unlogged watersheds and between wet and dry ecoregions. The underlying premise of the study was that headwater stream invertebrate communities are influenced by the type and quantity of allocthonous terrestrial plant material and by riparian shading. I hypothesized that the different riparian forest composition resulting

from different ecoregional climates, in the presence and absence of logging and post-logging forest succession, governs headwater stream invertebrate communities and leads to different invertebrate communities across four ecological conditions: dry-logged, dry-unlogged, wet-logged, and wet-unlogged.

1.3 Methods

Study sites

The study was conducted within the Wenatchee River subbasin located in the south central Cascade Range of Washington state (Fig. 1.1). Candidate streams (first and second order) were selected using topographic maps, aerial photos, and field reconnaissance for six major drainages: Mission Creek, Peshastin Creek, Icicle Creek, Nason Creek, Little Wenatchee River, and White River. Sample locations on each stream ranged in elevation from 561-1,341 m (Appendix 1.A). Six fishless headwater streams were selected from within each of four treatments for a total of 24 sites. These treatments were developed from two ecological subregions (wet and dry) and two land use categories (logged and unlogged) and will be referred to as: dry-logged (DL), dry-unlogged (DU), and wet-logged (WL), and wet-unlogged (WU). Ecological subregions were defined by Hessburg *et al.* (2000) using the TWINSPAN procedure to group watersheds according to higher order geology, landform features, potential vegetation types, and climate attributes. The ecological subregions (ESRs) selected for this study were ESR 4 – The Eastern Washington Cascades Moist & Cold Forests Subregion, and ESR 11 –The Eastern Washington Cascades Dry & Warm Forests Subregion, referred to here as wet and dry ecoregions respectively. Logged watersheds were characterized as

having been clear cut within the previous 30 years along one or both banks directly adjacent to study streams. Unlogged watersheds were characterized as having no evidence of timber harvest activity directly adjacent to study streams for at least 100 years. Time since last timber harvest was established using a combination of tree cores, logging scars on surrounding trees, and presence of logging roads and cut stumps.

Sampling design

Benthic invertebrate samples were collected once monthly in June, August, and October 2005, and May, July, and September 2006. Unseasonably warm weather during two weeks in May 2006 resulted in rapid snow melt and exceedingly high rates of stream discharge. Decreased visibility and increased stream depth made targeted riffle habitat difficult to locate, and sample nets had a tendency to back flush with high stream velocities; as a result, the validity of May samples was questionable, and these samples were excluded from further analysis. The 24 sites were sampled over a two to three week period during each monthly sampling. During these sampling periods, an equal proportion of sites from each of the four treatment categories were randomly sampled on any given day such that any temporal variability over the sampling period was evenly distributed among treatment categories.

Physical and chemical parameters

In each 100 m study reach, stream bank vegetation (herbaceous, bush, or tree), condition (stable or eroding), and angle (shallow, moderate, or undercut) were measured at 10 m intervals. Similarly, stream depth and substrate particle size (fine: <0.25 mm; sand: 0.25 - 2 mm; gravel: 2 -16 mm; pebble: 16 - 64 mm; cobble: 6.4 - 25 cm; or boulder

>25 cm) was measured at five equidistant points across the measured wetted width at each 10 m interval. Riparian canopy density and photosynthetically active radiation (PAR) were measured at 5 m increments once during peak leafout in 2005. Canopy cover was measured using a moosehorn canopy crown estimator (Garrison, 1949) to establish presence or absence of vegetation above the stream's center. Vegetation was identified to species to compare riparian coniferous and deciduous vegetation, specifically Sitka alder (*Alnus sinuate*), shown to be a highly nutritional plant species for terrestrial invertebrates (McComb, 1994). Photosynthetically active radiation (PAR) was measured using a LI 1400 PAR® meter. Water velocities were measured with an Intermountain Environmental® flow meter, and mean stream discharge was calculated for each from five recorded water velocities and depths measured at equidistant points across the measured wetted width of the sample stream. Where plunge pools or culverts existed, total stream discharge was estimated directly by measuring the time it took to fill a volumetric container over three trials. Stream temperature was measured every two hours with Onset TidBit® temperature loggers, and mean annual temperature was calculated using all data beginning January 15 and ending December 15, 2005. Conductivity, pH, and dissolved oxygen were measured at each sampling event with a 556 MPS YSI® meter. Percent slope was measured with a Sunto® clinometer at distances of no more than 25 m when vegetation density allowed.

Benthic invertebrates

A stratified random sampling design was used to select six sample locations of riffle habitat from a 100 m reach (two samples from each 33.3 m section). Benthic

invertebrates were sampled with a 250 μm D-net sampler from a 0.07 m^2 area (0.42 m^2 total) established by setting a wire boundary on the stream bottom. Substrate was then dislodged directly upstream of the D-net by hand. These six samples were combined into a single composite sample, and a standard 20 minutes was spent removing detritus and inorganic bed material to reduce sample volume. The composite sample was placed into a 500ml Whirl-Pak® bag and preserved in 100% ethanol. In the laboratory a minimum of 500 invertebrates was subsampled from each composite sample using a Folsom® plankton splitter. Large samples (too large to fit into Folsom® plankton splitter) were poured into a Caton® tray and 1/30 subsamples were removed and processed in the same manner as above. Insect taxa were identified to genus, and non-insect taxa were identified to order or class when reliable (Stewart & Stark, 1988; Merritt & Cummins, 1996; Wiggins, 1996). In addition, insect and non-insect taxa were assigned to a functional feeding group (Merritt & Cummins, 1996).

Periphyton

At each site, periphyton was sampled from natural rock substrate at each of the six invertebrate sampling locations; a single cobble size stone was haphazardly selected from riffle habitat, and periphyton were removed with a toothbrush from a 26.7 cm^2 area of its upper surface. These six individual samples were combined into a composite sample and stored in darkness in a labeled 100 ml Whirl-Pac® bag. In the laboratory each periphyton sample was immediately frozen. Chlorophyll *a* was extracted from half of each sample four weeks after collection using hot ethanol extraction (Sartory & Grobbelaar, 1984).

Chlorophyll *a* concentration was measured using spectrophotometric methods (Sartory & Grobbelaar, 1984).

Statistical analyses

Repeated measures analyses of covariance (ANCOVA) and Bonferroni corrected pair-wise comparisons were performed (SAS, version 9.1.2) to test for differences in invertebrate community assemblages among the main factors ecoregion and logging; month was the repeated measure, and mean annual stream temperature the covariate. The interaction term ecoregion × logging tested whether differences between logged and unlogged sites varied by ecoregion. Bonferroni corrected pair-wise comparisons were made at $\alpha = 0.0125$. Response variables exhibiting a non-normal distribution were log transformed to meet model assumptions with the exception of percentage related variables, which were arcsine square root transformed. Untransformed data are presented in all tables and figures unless otherwise stated. Analysis of variance (ANOVA) and Bonferroni corrected pair wise comparisons were performed using to test for differences in riparian vegetation and substrate composition among treatments categories.

Mean assemblage metric values were calculated for each treatment category by summing values of replicates within each treatment category and dividing by the number of replicates. Benthic density was defined as the number of individuals counted from a known area of stream bottom. Rarefied taxonomic richness was calculated at a sample size of 400 to account for differences among processed subsamples (Krebs & Brzustowski, 2007). It was not possible to rarefy samples to 500 individuals (the minimum sub-sample count) because some broken sections of segmented Oligochaeta

worms were identified as individual animals. Constraints of time and finances prevented further extraction of animals from those samples. Shannon-Weiner diversity was calculated as

$$H' = -\Sigma p_i \log(p_i)$$

Where p_i is the proportion of individuals in the ith taxon (Hauer & Resh, 1996). Percent abundance of Ephemeroptera, Plecoptera, Trichoptera (EPT) taxa was calculated as the numeric abundance of EPT taxa divided by the total numeric abundance of all taxa.

Cluster analysis

Cluster analysis (PC-ORD® version 4) was used to examine EPT composition similarity among the 24 study streams. EPT taxa present at fewer than 7% of sites were removed from the analysis resulting in 68 genera. Removal of these infrequent taxa helped reveal differences among treatment categories (McCune & Grace, 2002). The flexible beta method of clustering ($\beta = -1$) and a Sorensen (Bray-Curtis) distance measure were used in all cluster analysis.

1.4 Results

Environmental attributes

Mean annual stream temperatures ranged from 3.7-7.1 °C (Appendix 1.A) and were inversely related to elevation ($p < 0.001$, $r^2 = 0.78$) (Fig. 1.2). Mean stream velocity differed between months ($p < 0.001$) (Appendix 1.B) increasing over summer each season. Mean velocity was higher in unlogged watersheds ($p = 0.011$) (Table 1.1), and there was no detectable difference in total stream discharge among treatments.

Total riparian canopy cover ranged from 54-100% (Appendix 1.C) and after adjusting for aspect was higher in the wet ecoregion ($p = 0.041$) (Table 1.1; Fig. 1.3a). The proportion of coniferous cover ranged from 0-87% and was higher in unlogged watersheds ($p < 0.001$) and in the wet ecoregion ($p = 0.033$) (Table 1.1; Fig. 1.3b). Bonferroni corrected pair-wise comparisons indicated that the proportion of conifer cover between logged and unlogged watersheds differed significantly only in the wet ecoregion ($p < 0.001$), where unlogged watersheds had higher conifer densities in the wet ecoregion and logged watersheds had similar densities in both ecoregions (logging × ecoregion interaction, $p = 0.002$) (Fig. 1.3b). Deciduous cover ranged from 5-95% and was higher in logged watersheds ($p < 0.001$) (Table 1.1; Fig. 1.3c). The alder portion of this deciduous cover ranged from 0-73% and was also higher in logged watersheds ($p < 0.001$) (Fig. 1.3d). Pair-wise comparisons indicated significant differences in the proportion of both deciduous, and specifically alder, cover between logged and unlogged watersheds in the wet ecoregion only ($p < 0.001$ for both), where logged watersheds had higher densities. In the dry ecoregion, logged and unlogged watersheds had similar densities of both deciduous and alder cover (logging × ecoregion interaction, $p = 0.032$ and $p = 0.049$ respectively).

No significant difference in chlorophyll *a* biomass was detected among treatments. There was no detectible difference in substrate composition (% sand, gravel, pebbles, cobbles, or boulders) among treatment streams.

Invertebrate assemblages

Mean benthic density at individual sites ranged from 6,676-28,371 individuals per square meter (Appendix 1.D) and was higher in logged watersheds ($p = 0.009$) (Table 1.2; Fig. 1.4a). No difference in density was detected between sample years or between wet and dry ecoregions, and there was no logging × ecoregion interaction.

Mean % EPT at individual sites ranged from 19-57% (Appendix 1.D), and no difference was detected among treatments (Table 1.2). There was, however, a logging × ecoregion interaction ($p = 0.044$) (Fig. 1.4b) where the general pattern showed lower mean % EPT taxa in logged watersheds within the dry ecoregion and higher % EPT taxa in logged watersheds within the wet ecoregion. Mean % EPT was higher in 2005 than in 2006 ($p = 0.036$).

The dendrogram from cluster analysis of EPT composition was trimmed at three groups. This level of grouping retained roughly 38% of information and provided an interpretable summary of compositional similarities of EPT taxa among treatment categories (Fig. 1.5). All six sites comprising the DL treatment were grouped together based on the similarity of EPT composition. The dendrogram branch containing these sites was separated from the remaining sites early in the analysis indicating that the composition of EPT taxa in the DL treatment was different from that of the remaining treatment categories. All six sites comprising the WU treatment were also grouped together based on the similarity of EPT composition. Four of the six sites comprising the DU treatment were grouped together while only two of six sites comprising the WL treatment were grouped.

The total number of taxa identified across all sites was 126 with mean richness ranging from 26-38 (Appendix 1.D). No difference in mean taxa richness was detected among any treatment categories (Fig. 1.4c); however, mean richness was higher in 2005 than in 2006 ($p < 0.001$). Shannon-Weiner diversity ranged from 2.0-2.9 (Appendix 1.D) and no difference was detected among any treatment categories (Fig. 1.4d) or between sample years.

The general pattern of relative abundance for functional feeding groups was gatherers > shredders > predators > scrapers > filterers. This pattern was similar for all treatment categories except wet-unlogged where the proportion of predators exceeded shredders (Fig. 1.6). No differences in the density of individuals within functional feeding groups were detected between sample years. Gathering-collectors comprised 62-69% of the benthic community with higher densities in logged watersheds ($p = 0.014$) and in the dry ecoregion ($p = 0.023$) (Table 1.3; Fig. 1.7a). Chironomidae was the dominant gatherer in all treatments comprising 24-37% of that group. The remaining insect taxa comprised 17-21% of gatherer totals while the four non-insect taxa Ostracoda, Oligochaeta, Copepoda, and Planariidae comprised 43-47%. Shredders comprised 13-20% of the benthic community (Fig. 1.6) with higher densities in logged watersheds ($p < 0.001$) (Table 1.3; Fig. 1.7b). There was a significant logging × ecoregion interaction ($p = 0.006$) with unlogged watersheds having lower densities of shredders in the wet ecoregion and logged watersheds having similar densities in both ecoregions (Fig. 1.7b). Three stonefly genera *Zapada*, *Paracapnia*, and *Yoroperla* comprised 76-89% of shredder totals with *Zapada* dominant in unlogged watersheds and *Paracapnia* dominant

in logged watersheds. Predators comprised 11-15% of the benthic community (Fig. 1.6), and no difference in density was detected among treatments (Fig. 1.7c). Hydracarina mites were the dominant predator in all treatments representing 24-34% of predator abundance. Scrapers comprised 4-5% of benthic densities (Fig. 1.6), and no difference in density was detected among treatments (Fig. 1.7d). Filtering-collectors comprised 1-4% of benthic densities (Fig. 1.6) with higher densities in the wet ecoregion ($p = 0.003$) (Table 1.3; Fig. 1.7e). The net building caddisfly *Parapsyche* was the dominant filter feeder comprising 21-67% of filtering-collector totals.

Food and habitat

Numeric abundance of scrapers was not correlated with algal densities. Scrapers were the only functional group to show any correlation with substrate composition where density and the proportion of the community they represent were negatively correlated with percent sand substrate (Spearman's correlation coefficient = -0.31, $p = 0.004$ and -0.29, $p = 0.003$ respectively). Numeric abundance of shredders was positively correlated with the density of deciduous vegetation in August 2005 only (Spearman's correlation coefficient = 0.42, $p = 0.043$), but was correlated with alder density in both August 2005 and September 2006 (Spearman's correlation coefficient = 0.53, $p = 0.007$ for both).

1.5 Discussion

Results of this study suggest that differences in total invertebrate density and functional feeding group density between logged and unlogged watersheds varied by ecoregion and may be in response to differences in the quantity and type of potential food resources available. Differences detected in functional feeding group abundance suggest

that food resource availability originating from the terrestrial riparian forest may be a major contributor to differences in aquatic invertebrate community structure. It is generally assumed that shredders facilitate collector productivity by increasing the amount of fine particulate organic matter (FPOM) these animals depend on (Dieterich *et al.*, 1997). However, many studies supporting this relationship have been conducted under controlled conditions, and tests conducted in natural stream settings are lacking (Heard & Richardson, 1995). In this study we found that collector densities were not correlated with shredders densities. Shredders were found to be more abundant in wet-logged watersheds while no difference was seen in collector densities within the same treatment categories. This finding is in agreement with Usio *et al.* (2001) who found no correlation between shredding and collecting taxa, suggesting that collectors in this system are not particle-limited (Heard & Richardson, 1995). Benthic densities were higher in logged watersheds but only within the dry ecoregion. This was largely due to gathering-collectors who comprised 62-69% of benthic densities. Many biotic indices are designed around insect taxa (e.g., % EPT and % Chironomidae). While these indices have proven effective in describing invertebrate communities, four non-insect taxa Ostracoda, Oligochaeta, Copepoda, and Planariidae comprised 43-47% of gathering-collector totals in this study. This illustrates the importance of examining the non-insect component of aquatic ecosystems as a potential indicator of headwater condition. Lower taxa richness and % EPT in 2006 may have been due to differences in the seasonal hydrology of these two years. The range of taxa richness measured in this study (from 26-38) was lower than that previously measured in the Cascade Range by Li *et al.* (2001)

(from 48-66) and Whittier and Hughes (1988) (from 36-49). Several early season warming events in spring of 2006 caused severe flooding and channel scour in many of my sample streams. While it is possible that these high rates of discharge may have removed some taxa ill equipped to persist in such conditions, the lack of any difference in total numeric density suggests that remaining taxa were highly productive and flourished. In addition, the lack of any annual difference in the density of individuals within functional feeding groups suggests that the effects of such seasonal spates or resetting events, do not disproportionately influence food resource availability for any one feeding guild.

While no differences were detected in the percent of the community represented by EPT taxa among treatment categories, cluster analysis did detect differences in the composition of EPT taxa. The composition of EPT within replicates of the WL treatment least resembled one another, suggesting high variability in the composition of EPT taxa at wet-logged sites. In contrast, variability in EPT composition appears to be lower among sites within both DL and WU treatment categories as all six replicates resembled one another within each of these treatments. These results suggest that differences in community composition exist between ecoregions and may influence the resulting community structure following logging activities. More broadly it demonstrates the need to examine multiple metrics of invertebrate community structure consisting of both composite and compositional analysis.

Differences in benthic density were not related to differences in stream temperature. Mean annual temperatures were correlated with elevation, and treatment replicates were

evenly distributed across elevational gradients. Mean annual temperature was not correlated with percent canopy cover, nor did it significantly differ among treatments. Thus, temperature was unlikely to have influenced observed differences in invertebrate composite metrics or composition.

Past logging activity had no apparent influence on taxa richness but did influence community composition. Because logged sites were defined as those that had been harvested within the past 30 years, it may be that enough time had passed since logging to allow for habitats, and thus taxa richness, to return to pre-harvest conditions. Death & Winterbourn (1995) found that taxa richness peaked at high levels of habitat stability experiencing low disturbance, and taxa evenness peaked at intermediate levels of stability experiencing higher levels of disturbance. Because neither taxa richness nor Shannon-Weiner diversity were different among treatments it is possible that any shifts in habitat stability following logging had returned to pre-harvest conditions and that no differences existed between ecoregions. However, it is important to recognize that indices such as Shannon-Weiner diversity can be difficult to interpret because of the log-base calculations involved in generating these values. Even relative comparisons among sites may be difficult to interpret or explain. This index may be useful only when examined along side other metrics such as taxa richness, where potential differences in diversity among sites with similar richness suggest possible differences in the relative abundance of community members.

Total canopy cover was similar among treatments and may explain the similarities in algal density; however, any differences that may exist in primary production of algae may

have been obscured by the grazing activities of scraping taxa. All qualitative differences in riparian vegetation between logged and unlogged watersheds occurred only in the wet ecoregion. This may be due to differences in vegetation composition that existed prior to logging which were evident in unlogged watersheds. For example, the mean density of coniferous cover for dry-unlogged watersheds was 62% lower than in wet-unlogged watersheds. In contrast, mean density of deciduous cover for dry-unlogged watersheds was 55% higher than in wet-unlogged watersheds. Logging in the dry ecoregion may have resulted in a smaller shift in riparian vegetation composition (pre-existing high deciduous, low coniferous) than in the wet ecoregion where unlogged watersheds are dominated by coniferous cover. This less dramatic shift in deciduous vegetation in the dry ecoregion may explain why significant differences in shredder abundance were not observed in that ecoregion. The findings of (Power, 1992) support this assumption where she states, "plants have obvious primacy in food webs where primary productivity is a fundamental control of higher trophic levels". In addition to differences in functional structure, differences in taxonomic composition of shredders were also detected. The nemourid stonefly *Zapada* was the dominant shredder in unlogged sites while the capniid stonefly *Paracapnia* was dominant in logged sites. This is in contrast with findings of Wallace *et al.* (1986) who found that the recovery of functional structure following insecticide-induced disturbance was complete two years after treatment. It does, however, support their finding that differences in taxonomic composition within functional groups can exist long after the proportion of the community they represent has returned to pre-disturbance levels. The only other functional feeding group to show a

significant difference between logged and unlogged watersheds was gathering-collectors, which were more abundant in the dry ecoregion and were likely responsible for higher benthic densities in the dry-logged treatment. This may be due, in part, to higher organic particle densities in dry-logged watersheds. This organic material can originate from more erodible upland slopes typical of logged watersheds that cannot re-vegetate as quickly as wet-logged sites. This vegetation plays an important role in the retention of sediments (Middleton, 1999), and without it increased bare ground, stream bank erosion, and sediment deposition are likely to increase (Smith, 1993).

With larger volumes of suspended inorganic material in dry-logged watersheds, one would expect to see differences in substrate quality, in particular, increased percentages of sand. However, there were no detectable differences in the percent of any substrate particle class measured. The only detectable response of invertebrates to substrate habitat quality was a negative correlation between scrapers and sand substrate. This response by scrapers was most likely habitat and not food resource related because higher proportions of sand were not correlated with reduced algal densities.

1.6 Conclusions

In this study, differences between logged and unlogged watersheds did vary by ecoregion for gathering-collectors and shredders which composed the largest fraction of benthic communities. Although a retrospective study, these results suggest that the trajectory of invertebrate community succession following logging can be influenced by the ecoregion where logging has occurred.

Interpretation of observed biological patterns is likely dependant upon the scale at which those observations were made (Cooper *et al.*, 1998). Previous work examining patterns in invertebrate structure among ecoregions of the Pacific Northwest have done so among montane and non-montane regions such as the Willamette Valley, Columbia Basin, and high desert regions (Whittier *et al.*, 1988; Li *et al.*, 2001). While clear regional differences were seen among non-montane regions, regional differences among the montane streams were more subtle, and differences in invertebrate assemblages were indistinguishable at the taxonomic level of family (Whittier *et al.*, 1988). No other studies to my knowledge have attempted to examine these patterns among various ecoregions within the Cascade Range. This may be due, in part, to detailed classification of ecological subregions within the Cascades, such as those of Hessburg *et al.* (2000), not being readily available prior to 2000. This classification of ecological subregions was useful in identifying differences in invertebrate community structure within the Cascade Range and illustrates that more subtle differences in ecoregional characteristics inherent within this montane region are sufficient to elicit a response from stream biota.

This study has demonstrated that differences in the biological communities of headwater streams within the Cascade Range can be predicted using ecoregional classifications based on similar geology, landforms, climate, and potential vegetation composition. It may also be possible to make predictions about potential shifts in invertebrate community structure following future logging based on current riparian forest composition and predicted shifts in that composition depending upon ecoregion. These results suggest that ecoregional classifications could be developed for other

mountainous regions and tests for biological differences among classes could be conducted to make similar predictions. Use of such regional classifications and invertebrate metrics such as functional feeding group that appear to respond to differences in riparian forest composition could prove to be a valuable tool for forestry and fisheries managers in balancing viable and productive aquatic ecosystems with commercial timber harvest.

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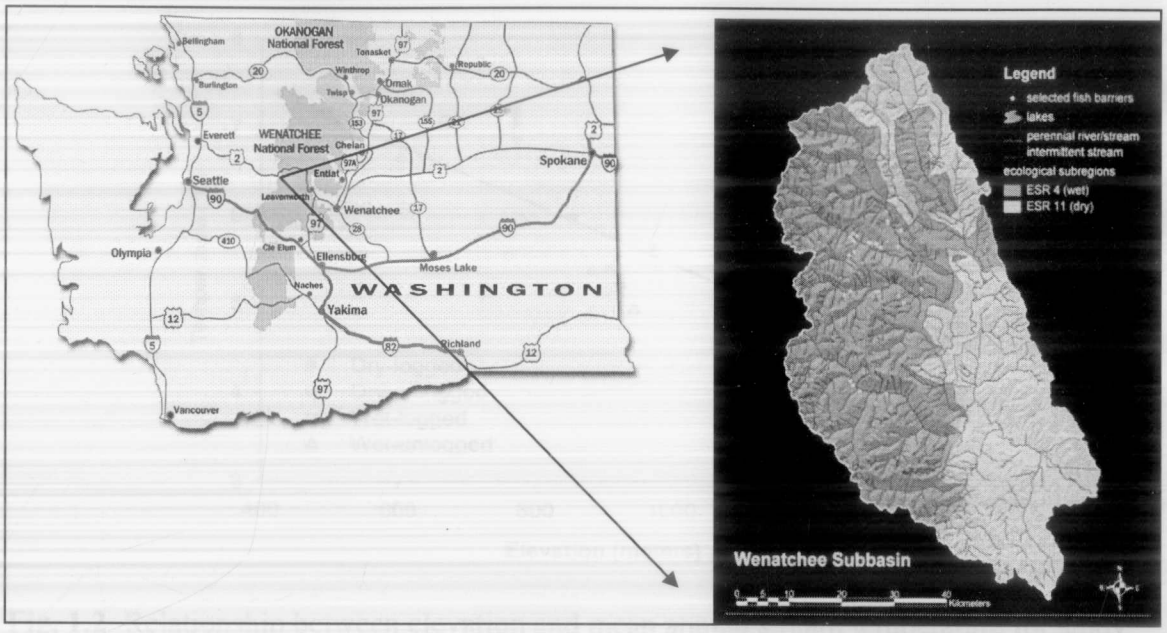

Fig. 1.1 Location of Wenatchee River subbasin.

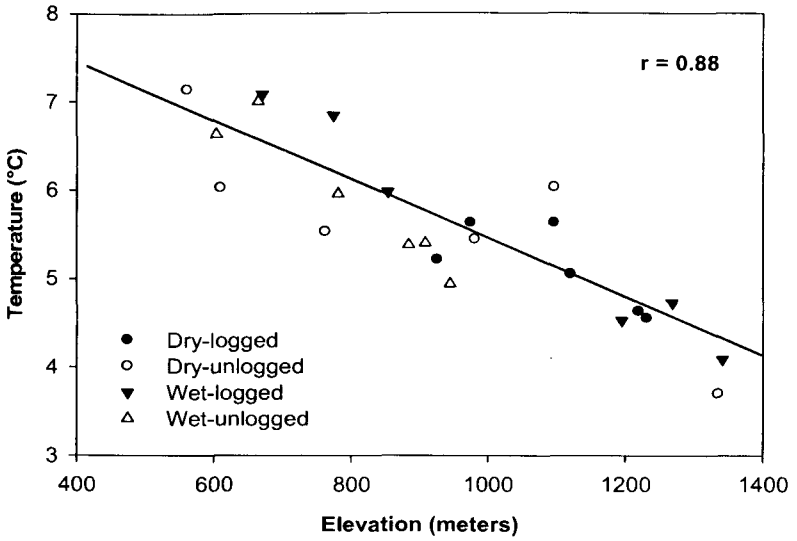

Fig. 1.2 Relationship between elevation and mean annual stream temperature for study sites in the Wenatchee River subbasin in 2005 and 2006.

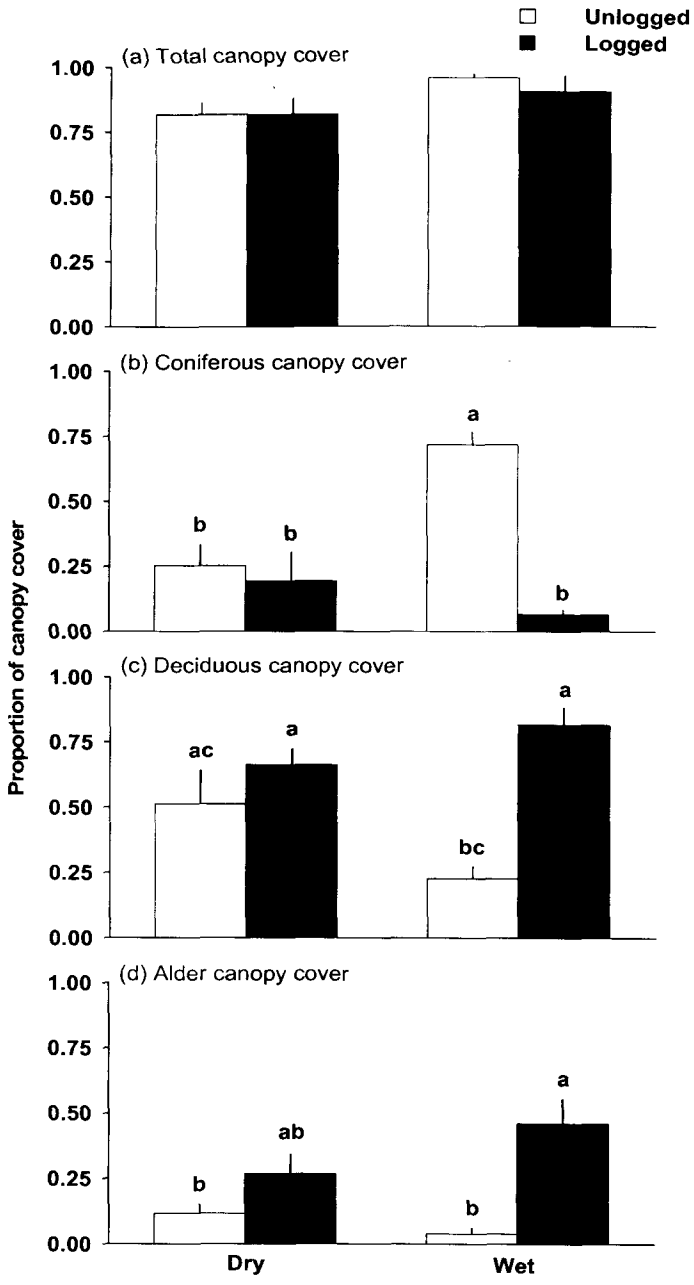

Fig. 1.3 Mean proportion (+1 SE) of riparian canopy cover for (a) total cover, (b) coniferous cover, (c) deciduous cover, and (d) alder cover in logged and unlogged watersheds, and dry and wet ecoregions. Means with the same letter were not statistically different based on Bonferroni's corrected *p*-values ($\alpha = 0.0125$); absence of letters indicate no significant differences among treatment categories.

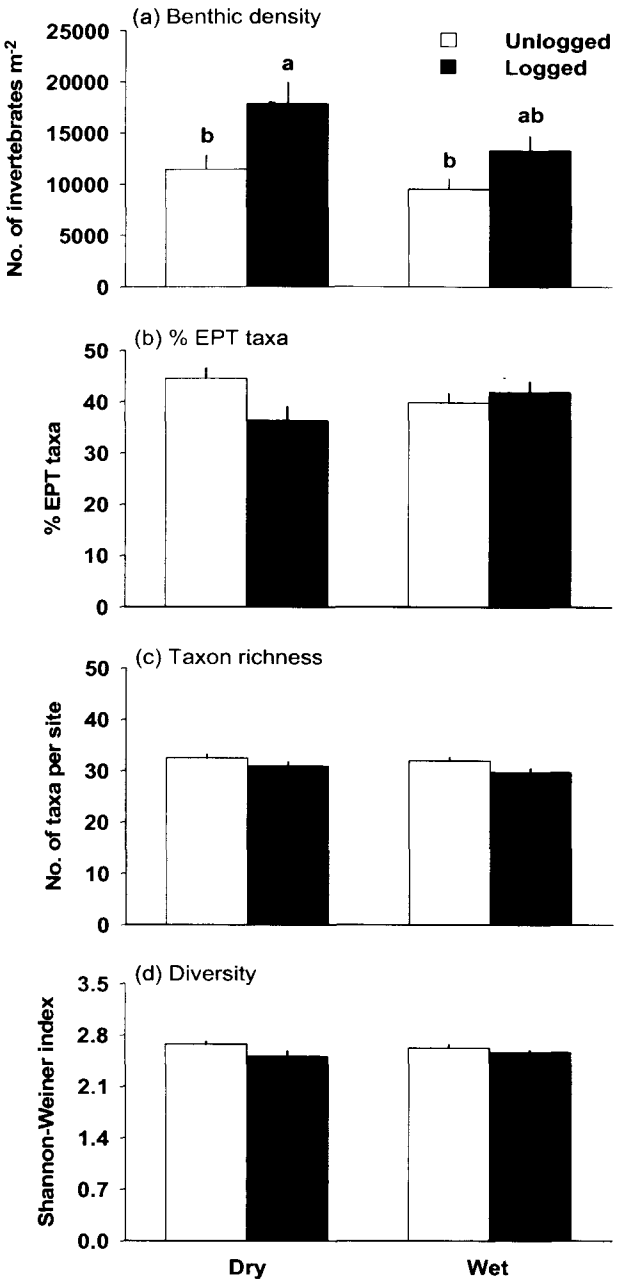

Fig. 1.4 Mean values (+1 SE) for (a) benthic density, (b) % EPT taxa, (c) taxa richness, and (d) Shannon-Weiner diversity in logged and unlogged watersheds, and dry and wet ecoregions. Means with the same letter were not statistically different based on Bonferroni's corrected *p*-values ($\alpha = 0.0125$); absence of letters indicate no significant differences among treatment categories.

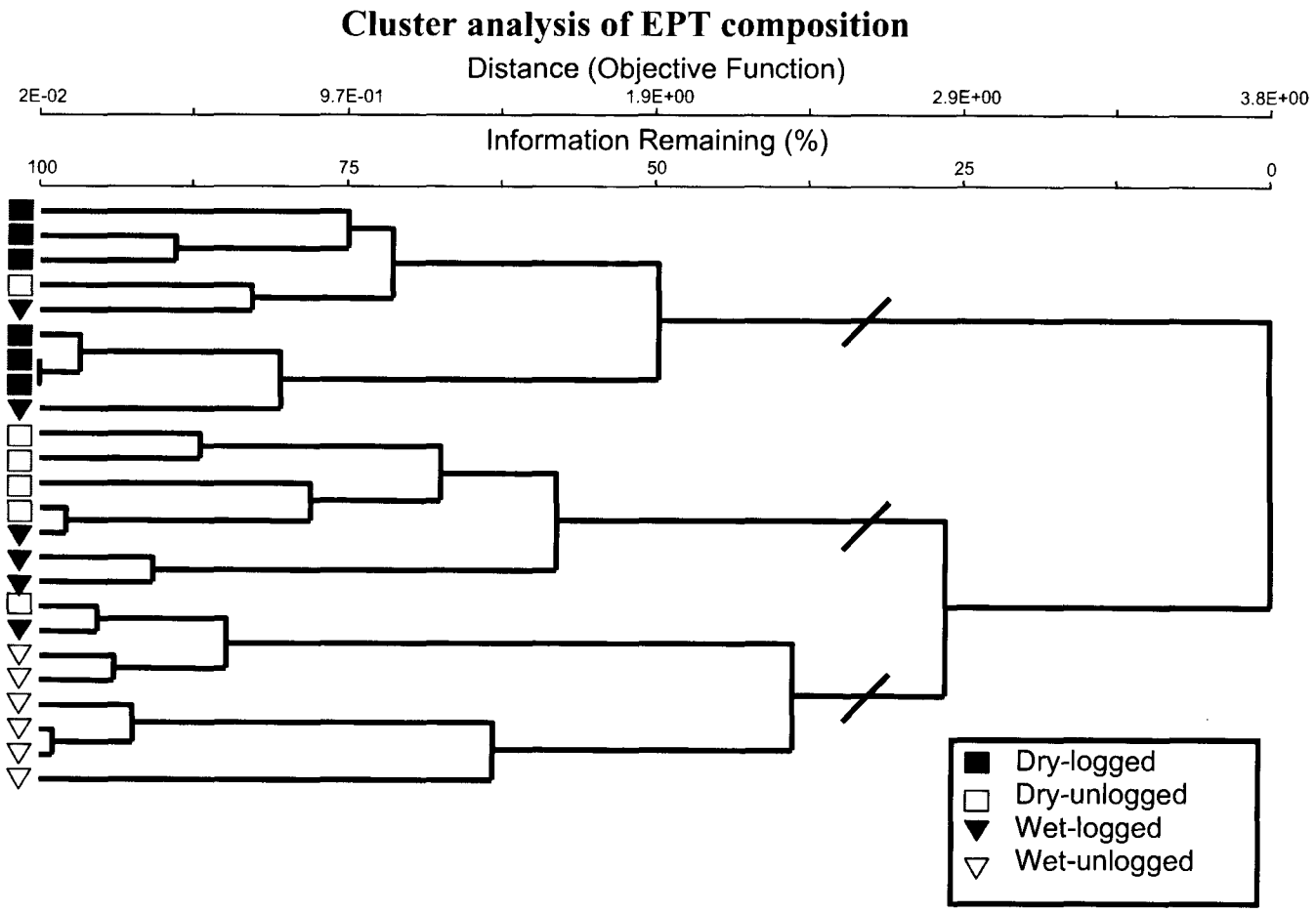

Fig. 1.5 Ephemeroptera, Plecoptera, Trichoptera (EPT) composition clustered by ecoregion and logging history. Flexible beta method of clustering ($\beta = -1$) and a Sorensen (Bray-Curtis) distance measure were used. / symbol represents location where the dendrogram was trimmed for analysis and similarities detected in EPT composition.

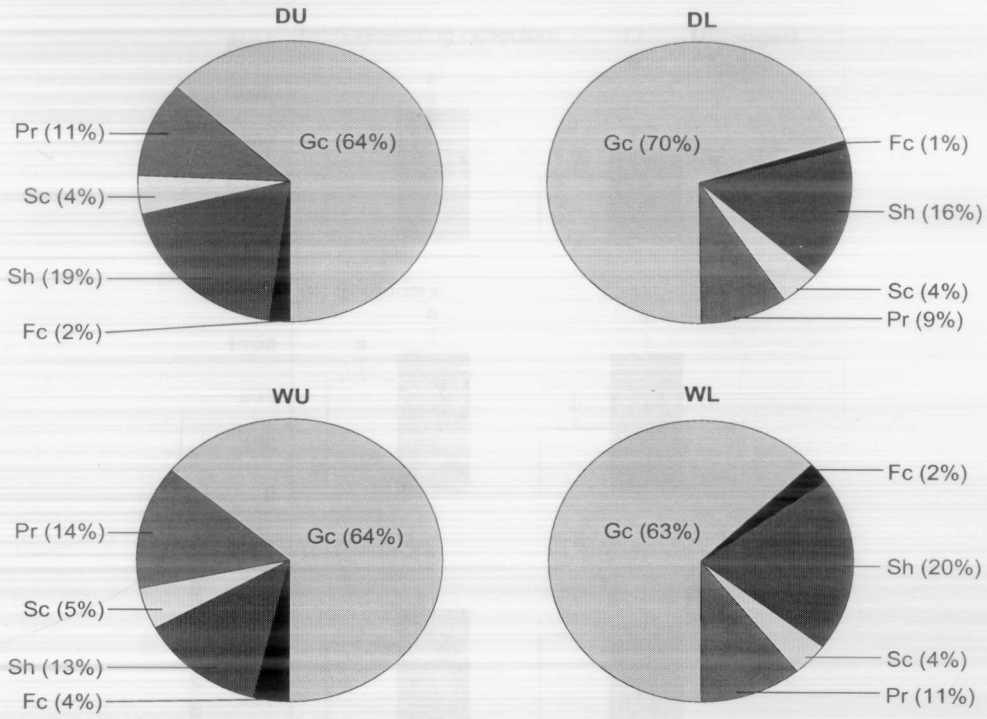

Fig. 1.6 Proportion of functional feeding groups, based on numeric abundance across four treatment categories. DU: dry-unlogged, DL: dry-logged, WU: wet-unlogged, WL: wet-logged, ■ filtering-collectors (Fc), ■ Scraper (Sc), □ Shredders (Sh), □ Predators (Pr), and □ Gathering-collectors (Gc).

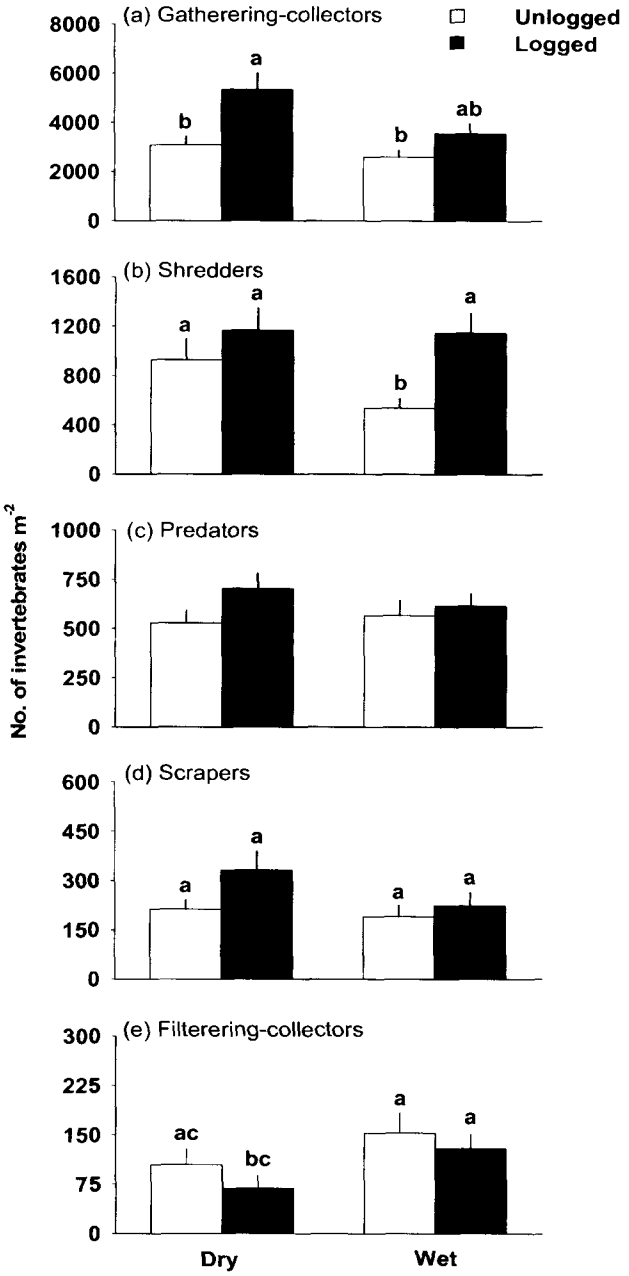

Fig. 1.7 Mean (+1 SE) abundance of individuals within functional feeding groups for (a) gathering-collectors, (b) shredders, (c) predators, (d) scrapers, and (e) filtering-collectors in logged and unlogged watersheds, and dry and wet ecoregions. Means with the same letter were not statistically different based on Bonferroni's corrected *p*-values ($\alpha = 0.0125$); absence of letters indicate no significant differences among treatment categories.

Table 1.1 Site characteristics and environmental attributes of study streams for each of four treatment categories. Standard errors are in parentheses.

Treatment Category	Mean elevation (m)	Mean channel width (cm)	Mean depth (cm)	Mean stream velocity (m/sec)	Mean stream discharge (L/sec)	Mean annual temperature (°C)	Canopy cover: Percent conifer	Canopy cover: Percent deciduous	Canopy cover: Percent alder	Canopy cover: Percent total cover
Dry-logged	1095	98.3	7.0	0.15	8.2	5.1	19.4	66.4	26.8	81.5
	(51)	(17.9)	(1.6)	(0.06)	(2.0)	(0.2)	(10.8)	(6.1)	(7.5)	(6.1)
Dry-unlogged	891	83.2	5.6	0.33	14.7	5.6	25.2	51.2	11.6	81.7
	(123)	(11.3)	(0.7)	(0.09)	(3.5)	(0.5)	(8.0)	(12.8)	(3.5)	(4.4)
Wet-logged	1017	101.0	6.0	0.16	9.1	5.5	6.6	81.7	46.1	90.2
	(116)	(20.7)	(1.5)	(0.05)	(2.8)	(0.5)	(1.8)	(6.6)	(9.4)	(6.1)
Wet-unlogged	798	111.3	7.7	0.16	13.8	5.9	71.7	22.7	3.9	95.4
	(57)	(5.0)	(0.8)	(0.03)	(3.2)	(0.3)	(4.8)	(4.6)	(2.2)	(1.5)

Table 1.2 Mean composite invertebrate assemblage characteristics for each of four treatment categories. Standard errors are in parentheses.

Treatment Category	Mean density (no. m^{-2})	Mean taxa richness	Mean % EPT individuals	Mean % dominant taxa	Mean Shannon diversity
Dry-logged	17515 (2689)	31.5 (1.0)	35.0 (4.2)	30.1 (4.5)	2.52 (0.12)
Dry-unlogged	11287 (1370)	33.8 (1.3)	45.3 (3.3)	23.3 (1.0)	2.71 (0.05)
Wet-logged	15202 (2412)	30.1 (1.3)	44.0 (3.1)	23.8 (1.6)	2.60 (0.06)
Wet-unlogged	9923 (855)	33.0 (1.1)	43.5 (1.6)	27.4 (3.0)	2.69 (0.08)

Table 1.3 Mean density (individuals m^{-2}) of functional feeding groups for each of four treatment categories. Standard errors are in parentheses.

Treatment Category	Filtering-collectors	Gathering-collectors	Predators	Scrapers	Shredders
Dry-logged	68 (19)	5321 (671)	703 (77)	332 (56)	1168 (176)
Dry-unlogged	105 (23)	3082 (351)	528 (64)	213 (27)	927 (170)
Wet-logged	129 (21)	3552 (397)	616 (61)	224 (39)	1146 (160)
Wet-unlogged	153 (30)	2593 (276)	566 (77)	190 (34)	535 (79)

1.8 Appendices.

Appendix 1.A

Study stream characteristics within the Wenatchee River Subbasin

Site	Treatment	Mean elevation (m)	Drainage aspect	Mean channel width (cm)	Mean depth (cm)	Median particle class	Mean stream velocity (m/sec)	Mean stream discharge (L/sec)	Mean annual temperature (°C)
Hansel 6	Dry-logged	1231	SE	49.2	3.8	sand	0.05	1.2	4.6
Hansel rock	Dry-logged	1219	SE	89.1	5.2	gravel	0.25	11.5	4.6
Scotty	Dry-logged	927	N	116.5	12.4	pebble	0.04	6.2	5.2
South Shaser	Dry-logged	975	NW	176	11.5	cobble	0.07	15.0	5.6
Trib Mid Shaser	Dry-logged	1097	N	85.7	3.0	pebble	0.39	9.7	5.6
Tronsen East	Dry-logged	1122	NW	73.4	6.4	pebble	0.12	5.7	5.0
Allen	Dry-unlogged	561	NE	122	8.8	pebble	0.23	24.8	7.1
Hansel 1	Dry-unlogged	1097	SE	108.2	6.8	pebble	0.29	21.3	6.0
King	Dry-unlogged	981	S	59	4.5	pebble	0.66	17.7	5.4
Mission Top	Dry-unlogged	762	SW	57.8	4.6	sand	0.13	3.4	5.5
Sand 1	Dry-unlogged	610	N	62.8	4.6	cobble	0.54	15.6	6.0
Tronsen 2	Dry-unlogged	1335	NW	89.2	4.7	pebble	0.13	5.3	3.7
Dirty Face	Wet-logged	853	SW	115.2	6.3	pebble	0.24	17.8	6.0
Fir	Wet-logged	671	S	61	3.7	gravel	0.07	1.6	7.1
Gill	Wet-logged	1341	NE	194.2	13.1	gravel	0.06	16.4	4.1
Hidden	Wet-logged	774	S	100.1	4.4	gravel	0.17	7.7	6.9
Roaring b	Wet-logged	1195	NE	78.2	3.7	gravel	0.06	1.8	4.5
Upper Trout	Wet-logged	1268	NW	57.4	4.6	pebble	0.35	9.2	4.7
Butcher East	Wet-unlogged	780	SW	118.7	6.7	pebble	0.24	19.1	6.0
End Ice 1	Wet-unlogged	884	NE	104.5	6.9	pebble	0.13	9.0	5.4
End Ice 3	Wet-unlogged	908	NE	102.8	5.7	pebble	0.17	10.0	5.4
Horse	Wet-unlogged	945	N	123.5	9.0	pebble	0.22	24.9	4.9
Sears	Wet-unlogged	604	NE	123.7	11.0	pebble	0.12	16.7	6.6
SLW	Wet-unlogged	664	NE	94.5	7.1	sand	0.05	3.4	7.0

Appendix 1.B

Mean velocity and discharge values for individual sites.

	Velocity (cm / second)				
	Jun-05	Aug-05	Oct-05	Jul-06	Sep-06
Hansel 6	10.6	2.1	0.8	12.6	3.0
Hansel Rock	44.2	20.3	10.4	54.2	9.0
Scotty	9.7	1.3	1.9	8.0	1.7
South Shaser	18.5	2.0	1.7	7.4	2.3
Trib Mid Shaser	38.5	na	na	na	na
Tronsen East	18.6	4.1	13.5	7.1	2.0
Allen	54.0	5.8	9.7	49.4	18.0
Hansel 1	52.8	17.8	17.0	39.3	17.0
King Canyon	121.3	42.5	34.7	76.4	63.9
Mission Top	26.3	7.8	4.3	16.1	6.8
Sand 1	69.7	79.5	13.6	58.8	21.0
Tronsen 2	13.7	2.2	22.4	1.7	na
Dirty Face	35.5	21.8	16.0	41.8	32.0
Fir	11.1	5.0	5.6	33.0	6.3
Gill	9.8	6.1	3.5	17.8	5.0
Hidden	24.2	11.0	17.0	23.5	9.9
Roaring	10.0	4.4	4.7	76.6	2.8
Upper Trout	34.7	56.8	12.6	na	na
Butcher East	34.3	23.9	14.1	85.7	46.0
End Ice 1	23.5	8.1	6.0	41.8	12.7
End Ice 3	31.7	12.1	6.9	47.6	8.1
Horse	46.8	11.9	8.4	40.8	8.6
Sears	22.8	8.5	5.6	18.9	5.2
SLW	6.3	2.1	7.0	10.8	3.1

Discharge (L / second)				
Jun-05	Aug-05	Oct-05	Jul-06	Sep-06
2.0	0.4	0.2	2.4	0.6
20.3	9.3	4.8	24.9	4.1
14.0	1.9	2.7	11.6	2.4
37.5	4.0	3.5	15.0	4.7
9.7	na	na	na	na
8.7	1.9	6.3	3.3	0.9
57.8	6.2	10.4	52.9	19.3
38.6	13.0	12.4	28.8	12.4
32.4	11.4	9.3	20.4	17.1
7.0	2.1	1.1	4.3	1.8
20.0	22.8	3.9	16.9	6.0
5.7	0.9	9.3	0.7	na
25.8	15.9	11.7	30.4	23.3
2.5	1.1	1.3	7.4	1.4
24.8	15.4	8.9	45.2	12.8
10.8	4.9	7.6	10.4	4.4
2.8	1.3	1.4	21.9	0.8
9.2	15.1	3.3	na	na
27.2	19.0	11.2	68.0	36.5
16.9	5.8	4.3	30.0	9.1
18.7	7.1	4.1	28.0	4.8
52.2	13.2	9.4	45.6	9.6
30.9	11.5	7.6	25.6	7.0
4.2	1.4	4.6	7.2	2.0

Appendix 1.C

Percent riparian forest cover for individual sites.

Site name	Treatment category	Conifer	Deciduous	Alder	Total canopy cover
Hansel 6	dry-logged	0	49	24	54
Hansel Rock	dry-logged	6	80	52	91
Scotty	dry-logged	2	85	21	93
South Shaser	dry-logged	8	67	7	88
Trib Mid Shaser	dry-logged	68	68	46	75
Tronsen East	dry-logged	33	49	10	89
Allen	dry-unlogged	45	40	1	89
Hansel 1	dry-unlogged	1	77	10	78
King	dry-unlogged	29	38	18	69
Mission Top	dry-unlogged	2	94	25	98
Sand 1	dry-unlogged	29	54	11	83
Tronsen 2	dry-unlogged	45	5	5	74
Dirty Face	Wet-logged	14	63	9	83
Fir	Wet-logged	5	95	32	100
Gill	Wet-logged	3	59	59	63
Hidden	Wet-logged	8	92	45	100
Roaring b	Wet-logged	6	88	73	98
Upper Trout Trib	Wet-logged	2	94	59	98
Butcher East	Wet-unlogged	68	22	2	97
End Ice 1	Wet-unlogged	70	22	0	91
End Ice 3	Wet-unlogged	85	12	0	98
Horse	Wet-unlogged	57	43	13	90
Sears	Wet-unlogged	64	26	0	97
SLW	Wet-unlogged	87	12	7	99

Appendix 1.D

Benthic invertebrate assemblage characteristics by site.

Site name	Mean density (no. m^{-2})	Mean taxa richness	Mean % EPT individuals	Mean % dominant individuals	Mean Shannon diversity
Hansel 6	18313	28	19	49.7	2.02
Hansel rock	11643	30	31	31.9	2.44
Scotty	28371	31	32	33.2	2.41
South Shaser	13893	33	47	20.1	2.85
Trib Mid Shaser	21363	33	37	23.5	2.72
Tronsen East	11507	34	45	22.1	2.68
Allen	13450	32	45	26.8	2.64
Hansel 1	10229	37	42	25.0	2.86
King	17010	33	57	20.1	2.78
Mission Top	7809	38	32	23.5	2.77
Sand 1	9510	33	48	22.9	2.71
Tronsen 2	9715	30	48	21.6	2.53
Dirty Face	6676	32	51	26.5	2.52
Fir	15047	35	38	29.9	2.68
Gill	24152	30	52	22.2	2.69
Hidden	14280	30	49	19.8	2.79
Roaring b	12308	27	38	20.0	2.52
Upper Trout	18747	26	36	24.2	2.41
Butcher East	10997	36	40	33.7	2.52
End Ice 1	9721	33	44	18.8	2.90
End Ice 3	13318	35	44	24.8	2.85
Horse	7982	31	45	19.5	2.78
Sears	9935	34	49	32.2	2.71
SLW	7585	28	38	35.5	2.40

CHAPTER 2

Do drift assemblages reflect benthic communities in headwater streams?

2.1 Abstract

Sampling and analyzing invertebrate drift for assessing stream condition, compared to traditional benthic sampling, has received little attention. Drifting invertebrates can be easier, quicker, and less expensive to sample and process than benthic invertebrates and may hold promise for assessing stream condition. The objective of this study was to determine if drift samples might serve as a replacement for benthic samples in assessing the relative condition of streams. Benthic and drifting invertebrates were sampled in 24 headwater streams within logged and unlogged watersheds from both wet and dry ecoregions of the eastern Cascade Range; they were analyzed for relationships between eleven community metrics, similarities in community composition, and differences in drift propensity. Benthic assemblage metrics differed between ecoregions and logging histories for six of eleven metrics examined compared to one of eleven metrics examined for drift. Significant relationships between the benthos and drift were not consistently detected across ecoregions. A negative relationship was detected between total benthic density and drift propensity suggesting that potentially less productive systems with lower benthic densities also have higher rates of emigration. Drift samples did not appear to reflect benthic communities and thus, would be an inadequate replacement for benthic sampling in assessing headwater stream condition.

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2.2 Introduction

Sampling aquatic organisms to biologically assess stream condition has a long history (Kolkwitz & Marsson, 1908; Hilsenhoff, 1977, 1998) and is employed by many state and federal agencies (U.S. EPA, 2004). Various aquatic organisms (e.g., bacteria, periphyton, invertebrates, fish) have been used in biological assessment; however, invertebrates are most often recommended Hellawell (1986). Invertebrates are numerous, and easy to collect, and their communities contain many species with various degrees of sensitivity to environmental stressors covering a wide spectrum of potential effects (Resh *et al.*, 1996).

Measures of invertebrate community structure can be broken into five broad categories (Resh & Jackson, 1993) including: enumerations, which range from counts of all organisms to relative abundance of taxonomic units; richness, which describes the number of distinct, specific taxonomic units (e.g., family); diversity, which combines richness and enumerations in a summary statistic; biotic indices, which use pre-established water-quality tolerance values for taxonomic units; and functional feeding groups, which examine differences in the relative abundance of invertebrates with similar food acquisition behaviors.

Secondary benthic production can be a useful measure of relative stream condition and is rarely measured directly, but rather inferred from instantaneous samples of stream biota (Woodcock & Huryn, 2007). This secondary production represents an estimate of the flow of energy through populations and communities (Benke *et al.*, 1999) including exchange between aquatic and terrestrial ecosystems (Ballinger & Lake, 2006) and transport of energy to downstream consumers via invertebrate drift (Wipfli *et al.*, 2007).

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Secondary benthic production can be a useful measure of relative stream condition and is rarely measured directly, but rather inferred from instantaneous samples of stream biota (Woodcock & Huryn, 2007). This secondary production represents an estimate of the flow of energy through populations and communities (Benke *et al.*, 1999) including exchange between aquatic and terrestrial ecosystems (Ballinger & Lake, 2006) and transport of energy to downstream consumers via invertebrate drift (Wipfli *et al.*, 2007).

The transport of this energy may be impaired with increasing stress to aquatic systems due to anthropogenic disturbance (Woodcock & Huryn, 2007). When estimating productivity or impairment of productivity, it is important to examine the balance of individuals filling various functional roles (e.g., breaking down of course organic matter, algae consumption). This is because it is possible to have reduced energy flow and overall ecosystem impairment, despite increases in the production of tolerant taxa (Woodcock & Huryn, 2007).

Alteration of stream habitat by logging can influence the structure and function of invertebrate communities by reducing flow of energy, increasing sedimentation (Death *et al.*, 2003), elevating temperatures (Quinn *et al.*, 2004), and reducing large woody debris (Richardson *et al.*, 2005). Logging also can promote the growth of deciduous riparian vegetation such as alder which has been shown to increase the abundance of drifting invertebrates (Wipfli & Musslewhite, 2004), including shredders and gatherers that rely on inputs of this allochthonous food resource.

Because physical habitat can influence the relative abundance of invertebrate communities, biotic indices are typically generated for particular regions with similar environmental attributes. The vast majority of biomonitoring methods utilize benthic sampling, in part, for comparability with other studies, and because the sampling techniques and data analysis are well developed (Resh *et al.*, 1996). Wang *et al.* (2006) compared invertebrate communities from riffle and snag habitats to evaluate the influence of habitat-specific sampling on community assemblage measures and biotic indices. They concluded that invertebrate indices developed for benthic riffle habitats

should not be used for snag samples to assess stream impairment and recommended that biotic indices be developed specifically for snag habitats. Previous work has shown a positive relationship between benthic densities and drift densities, suggesting that mean drift rate might be used as an index of a stream's productive capacity (Waters, 1961; Siler *et al.*, 2001), which may reflect its overall condition. If such a relationship exists between benthic and drifting invertebrates, it is possible that differences in benthic densities among streams may be reflected in the drift assemblage. This point notwithstanding, it is also possible that a relationship exists between the benthos and the drift based not on their total densities, but rather the composition of individuals within each of these assemblages. An invertebrates habitat preference and mode of locomotion can be described as habit (Merritt & Cummins, 1996). Differences in habit are likely to influence relationships between the benthic and drifting assemblage metrics where highly mobile taxa such as Baetidae mayflies may show stronger similarities between the benthos and drift compared to more sessile taxa such as Simuliidae Diptera. As differences between benthic and drift samples have received limited attention, the purpose of this work was to compare the influence of these techniques of assemblage-specific sampling on community measures and biotic indices.

Benthic and drifting invertebrates were sampled in two ecoregions and two levels of logging to determine 1) the comparability of drift assemblages and benthic communities under each of these conditions, and 2) the community metrics that detect potential differences in each of these assemblages between ecoregions and levels of logging. The

broader goal was to determine whether drift samples may be an effective proxy for benthic samples in assessing the biological condition of headwater streams.

2.3 Methods

Study sites

The study was conducted within the Wenatchee River subbasin located in the south central Cascade Range of Washington state (Fig. 1.1). Candidate streams (first and second order) were selected using topographic maps, aerial photos, and field reconnaissance for six major drainages: Mission Creek, Peshastin Creek, Icicle Creek, Nason Creek, Little Wenatchee River, and White River. Sample locations on each stream ranged in elevation from 561-1,341 m (Appendix 1.A). Six fishless headwater streams were selected from within each of four treatments for a total of 24 sites. These treatments were developed from two ecological subregions (wet and dry) and two land use categories (logged and unlogged) and will be referred to as: dry-logged (DL), dry-unlogged (DU), and wet-logged (WL), and wet-unlogged (WU). Ecological subregions were defined by Hessburg *et al.* (2000) using the TWINSPAN procedure to group watersheds according to higher order geology, landform features, potential vegetation types, and climate attributes. The ecological subregions (ESRs) selected for this study were ESR 4 – The Eastern Washington Cascades Moist & Cold Forests Subregion, and ESR 11 –The Eastern Washington Cascades Dry & Warm Forests Subregion, referred to here as wet and dry ecoregions respectively. Logged watersheds were characterized as having been clear cut within the previous 30 years along one or both banks directly adjacent to study streams. Unlogged watersheds were characterized as having no

evidence of timber harvest activity directly adjacent to study streams for at least 100 years. Time since last timber harvest was established using a combination of tree cores, logging scars on surrounding trees, and presence of logging roads and cut stumps.

Sampling design

Benthic and drift samples were collected once monthly in August, and October 2005. The 24 sites were sampled over a two to three week period during each sampling. During these sampling periods, equal proportions of sites from each of the four treatment categories were sampled on any given day such that any temporal variability over the sampling period was evenly distributed across treatments. Logistical constraints in October 2005 allowed only for the sampling of four replicate streams in each treatment category.

Physical and chemical parameters

In each 100 m study reach, stream depth and substrate particle class (fine: <0.25 mm; sand: 0.25-2 mm; gravel: 2-16 mm; pebble: 16-64 mm; cobble: 6.4-25 cm; or boulder >25cm) was measured at five equidistant points across the measured wetted width at each 10 m interval. Temperature was measured at each site every two hours with Onset TidBit® temperature loggers. Conductivity, pH, and dissolved oxygen were measured at each sampling event with a 556 MPS YSI® meter. Percent slope was measured with a Sunto® clinometer at distances of no more than 25 m when vegetation density allowed. Total stream discharge was calculated from five recorded water velocities and depths measured at equidistant points across the measured wetted width of the sample stream. All measurements were made at the time of net placement and retrieval. Where plunge

pools or culverts existed, total stream discharge was measured directly by averaging the time it took to fill a volumetric container over three trials.

Drifting invertebrates

Invertebrate drift samples were collected over a 24 hr period using drift nets (mesh size 250 μm) mounted on 0.3 m section of square (6.4 cm x 6.4 cm) plastic pipe. Nets were positioned in the stream's thalweg to maximize flow capture. The total volume of water sampled was calculated from measured depths and water velocities at the mouth of the sample pipe using an Intermountain Environmental® flow meter. Once each drift net was removed from the stream, its contents were rinsed to the bottom and the entire net was placed in a Zip-Lock® bag for transport. In the laboratory, sample contents were removed from each net and preserved in 100% ethanol.

After a sample was passed through nested 1mm and 250 μm sieves, all invertebrates retained on the 1 mm sieve were counted and identified. A minimum count of 300 invertebrates retained on the 250 μm sieve were identified. When samples retained on the 250 μm sieve exceeded 600 invertebrates, they were sub-sampled to achieve the 300-count estimate (1/8 to 1/64 of whole sample) using a Folsom® plankton splitter, prior to identification. All terrestrial invertebrates were removed and not analyzed.

Benthic invertebrates

Benthic invertebrate samples were collected immediately after drift nets were retrieved whenever possible. In some cases, benthic samples were collected many days after drift samples. A stratified random sampling design was used to select six sample locations of riffle habitat from a 100 m reach (two samples from each 33.3 m section)

located above the point of drift sampling. Benthic invertebrates were sampled with a 250 μm D-net sampler from a 0.07 m^2 area (0.42 m^2 total) established by setting a wire boundary on the stream bottom. Substrate was then dislodged directly upstream of the D-net by hand. These six samples were combined into a single composite sample, and a standard 20 minutes was spent removing detritus and inorganic bed material to conserve storage space. The composite sample was placed into a 500 ml Whirl-Pak® bag and preserved in 100% ethanol. In the laboratory, a minimum of 500 invertebrates was subsampled from each composite sample using a Folsom® plankton splitter. Large samples (too large to fit into Folsom® plankton splitter) were poured into a Caton® tray, and 1/30 subsamples were removed and processed in the same manner as above. Insect taxa from drift and benthic samples were identified to family, and non-insect taxa were identified to order or class when reliable (Stewart & Stark, 1988; Merritt & Cummins, 1996; Wiggins, 1996). In addition, each taxon identified was assigned to a functional feeding group (Merritt & Cummins, 1996).

Statistical analyses

Linear regression was used to examine the relationship between benthic and drift samples for six invertebrate assemblage measures: % EPT taxa, Hilsenhoff's family-level biotic index, family-level richness, numeric density, % dominant taxa, and Shannon-Wiener diversity. Data with non-normal distributions were log transformed in the case of count data and arcsine square root transformed in the case of percentage data. Repeated measures analyses of covariance (ANCOVA) were performed (SAS® version 9.1.2) to test for differences in the mean values of these invertebrate assemblage metrics among

ecoregions and logging treatments; month was the repeated measure and mean annual stream temperature the covariate. In addition, ANCOVA was performed to test for differences in numeric abundance of individuals within five functional feeding groups and community wide drift propensity, which represents the *per capita* rate of emigration from the benthos to the drift.

Mean assemblage metric values were calculated for each treatment category by summing values of replicates within each treatment category and dividing by the number of replicates. Percent abundance of Ephemeroptera, Plecoptera, Trichoptera (EPT) taxa was calculated as the numeric abundance of EPT individuals divided by the total numeric abundance of all taxa. Hilsenhoff's family-level biotic index (HFBI), where lower values indicate higher water quality, was calculated as

$$HFBI = 1/N \Sigma n_i t_i$$

where n_i is the number of individuals in a family; t_i, the tolerance score for the family (Hilsenhoff, 1988); and N, the total number of individuals in the sample. Rarefied family richness was calculated at a sample size of 400 individuals for benthic samples and 100 individuals for drift samples to account for differences in total individuals identified among processed subsamples (Krebs & Brzustowski, 2007). It was not possible to rarefy benthic samples to 500 individuals (the minimum sub-sample count) because once identified, it was realized that some broken sections of segmented Oligochaeta worms had been counted as individual animals. Constraints of time and finances prevented further extraction of animals from those samples. Drift samples were rarefied to 100, roughly the size of our smallest sample. Benthic density was calculated as the number of

individuals counted from a known area of stream bottom. Drift density was calculated as the number of individuals collected per cubic meter of water sampled over a 24 hr period. For both benthic and drift assemblages, % dominant taxa was calculated as the numeric abundance of the most abundant taxa divided by the total abundance. Shannon-Weiner diversity was calculated as

$$H' = -\Sigma p_i \log(p_i)$$

where p_i is the proportion of individuals in the *i*th taxon (Hauer & Resh, 1996). Drift propensity represents the *per capita* rate of entry into the drift from the benthos and is calculated by dividing drift density by benthic density. Examining drift propensity eliminates the possibility that observed patterns in drift rate simply reflect variations in benthic density (McIntosh *et al.*, 2002). Increases in the *per capita* rate of emigration from the benthos can identify areas of potentially inadequate habitat or insufficient resources to support the benthic standing stock (Siler *et al.*, 2001). Functional feeding group composition was examined for % shredders, scrapers, gathering-collectors, filtering-collectors, and predators (Merritt & Cummins, 1996). Each was determined by dividing the number of individuals within a functional group by the total number of invertebrates.

Cluster analysis

Cluster analysis (PC-ORD® version 4) was used to examine the similarity of invertebrate community composition among the 24 study streams and between benthic and drift assemblages. This analysis included benthic and drift assemblages in order to establish whether similarities detected among treatment categories in benthic

assemblages were also seen in drift assemblages. The proportion of families present at each site was calculated for both assemblages. Forty one families were analyzed after families present at fewer than 7% of sites were removed. Removal of these infrequent families better revealed differences among treatment categories (McCune & Grace, 2002). The flexible beta method of clustering ($\beta = -1$) and a Sorensen (Bray-Curtis) distance measure were used.

2.4 Results

Invertebrate densities

Though a negative relationship was detected between community wide drift propensity and benthic density ($r^2 = 0.15$, $p = 0.003$) (Fig. 2.1), no relationship between benthic and drift density was detected among any treatment categories for numeric density (Fig. 2.2a & b). Benthic densities ranged from 7,819-38,802 individuals m^{-2} (Appendix 2.A) and were higher in logged watersheds ($p = 0.011$) (Table 2.1; Appendix 2.B). Mean drift densities ranged from 1.1-32.9 individuals m^{-3} of water (Appendix 2.C) with no differences detected among any treatment categories. No difference in drift propensity was detected among any treatment categories.

Percent EPT and Hilsenhoff's Index

When all data were examined and no distinction was made among treatment categories, a positive relationship between the benthos and drift was detected for % EPT taxa ($r^2 = 0.15$, $p = 0.02$) (Fig. 2.3a) and for HFBI ($R^2 = 0.24$, $p = 0.002$) (Fig. 2.3b). Mean % EPT ranged from 15-56% for benthic assemblages (Appendix 2.A) and 23-75% for drifting assemblages (Appendix 2.C). When individual treatment categories were

considered, a positive correlation between the benthos and the drift was detected for % EPT in both logged and unlogged watersheds of the dry ecoregion sites, (DL treatment $r^2 = 0.60$, $p = 0.014$; DU treatment $r^2 = 0.52$, $p = 0.028$) (Fig. 2.4a). Taxonomic membership of the EPT group was similar in wet and dry ecoregions with 21 of 24 benthic taxa found in the drift of the wet ecoregion and 22 of 24 benthic taxa found in the drift of the dry ecoregion. ANCOVA detected no differences among any treatment categories for mean % EPT in either benthic or drift assemblages. A significant interaction between logging and ecoregion was detected ($p = 0.021$) for the benthic assemblage, but no clear pattern could be discerned.

Mean HFBI values ranged from 2.2-4.7 for benthic assemblages (Appendix 2.A) and 1.0-4.6 for drifting assemblages (Appendix 2.C) suggesting that water quality, as measured by this index (Hilsenhoff, 1988), ranges from good (4.26-5.00) to excellent (0.00-3.75) when either benthic or drift samples are used. Overall, drift samples suggested a higher mean water quality then did benthic samples (Fig. 2.3b) as the majority of data points fall below the theoretical 1:1 line. Benthic and drift samples classified 57% of sites into the same water quality class of "excellent". Drift samples classified 16% of sites into one class higher than benthic samples, 16% of sites into two classes higher, and 5% of sites into three classes higher; drift samples also classified 5% of sites into one class lower than benthic samples. When individual treatment categories were considered, a positive correlation was detected between benthic and drift samples in only the DL treatment ($r^2 = 0.49$, $p = 0.035$) (Fig. 2.5a). ANCOVA detected no

differences in HFBI among any treatment categories for either benthic or drift assemblages.

Family-level richness

Mean rarefied family-level richness ranged from 21-33 families for benthic assemblages (Appendix 2.A) and 9-19 families for drifting assemblages (Appendix 2.C). No correlation between the benthos and drift was detected for rarefied family-level richness (Fig. 2.3c), nor were any significant differences in this metric detected among any treatment categories.

Shannon-Weiner diversity and % dominant taxa

No correlation between the benthos and drift was detected for Shannon-Weiner diversity or % dominant taxa, nor were any significant differences in these metrics detected among any treatment categories.

Cluster analysis

Cluster analysis of benthic composition suggests that similarities may exist among treatment categories. When the dendrogram of benthic composition was trimmed at two groups (40% of information remaining) there appeared to be a pattern of differing benthic composition between dry and wet ecoregions with 67% of dry sites clustering out together and 75% of wet sites clustering out together (Fig. 2.6a). At this same trim point, there also was a separation of logged and unlogged sites based on composition with 67% of logged sites clustering out together and 75% of unlogged sites clustering out together (Fig. 2.6a). No clear pattern was seen in the cluster analysis of drift composition when

all taxa were considered. No clear patterns were seen in the cluster analysis of EPT composition for either benthic or drifting assemblages (Fig. 2.7a & b).

Functional feeding group densities

Scrapers were the only functional feeding group in the drift assemblage that differed significantly among treatment categories. Mean drifting scraper densities ranged from 0.01-0.56 individuals m^{-3} water (Appendix 2.D) and were more abundant in the dry ecoregion ($p = 0.001$) (Table 2.2). In contrast, no differences were detected among any treatment categories in mean benthic scraper densities (from 24-956 individuals m^{-2}, Appendix 2.E).

Mean drifting filtering-collector densities ranged from 0.00-0.21 individuals m^{-3} water (Appendix 2.D), and no differences in mean densities were detected among any treatment categories (Table 2.2). In contrast, mean benthic filtering-collector densities (0-388 individuals m^{-2}, Appendix 2.E) were higher in the wet ecoregion ($p = 0.008$) (Table 2.2; Appendix 2.B).

No differences were detected in mean drifting gathering-collector densities (0.3-18.1 individuals m^{-3} water, Appendix 2.D) among any treatment categories. Mean benthic gathering-collector densities (2,032-12,620 individuals m^{-2}, Appendix 2.E) were more abundant in logged watersheds ($p = 0.04$) (Table 2.2; Appendix 2.B).

No differences were detected in drifting shredder densities (0.12-2.07 individuals m^{-3} water, Appendix 2.D) among any treatment categories. Benthic shredder densities (416-2,784 individuals m^{-2}, Appendix 2.E) were more abundant in logged watersheds ($p =$

0.048) but only in the wet ecoregion (logging × ecoregion interaction, $p = 0.035$) (Table 2.2; Appendix 2.B).

Percent dominant taxa and Shannon-Wiener Diversity

Percent dominant taxa ranged from 16-46% for benthic assemblages (Appendix 2.A) and 17-67% for drift assemblages (Appendix 2.C). No relationship between the benthos and drift was detected for % dominant taxa among any treatment categories. Analysis of covariance detected higher percentages of dominant taxa in the benthos for the dry ecoregion ($p = 0.04$), but no differences were found among any treatment categories for drift assemblages. The benthos was dominated by Chironomidae in all treatment categories followed by Oligochaeta in the WU and DU treatments and Capniidae in the WL and DL treatments. Four taxa comprised greater than 50% of all drift community totals: Ostracoda, Chironomidae, Baetidae, and Hydracarina. Ostracods showed the greatest ecoregional difference of any taxa, comprising 40 and 42% of community totals in the DU and DL treatments, respectively, compared to 12 and 11% in the WU and WL treatments.

Shannon-Wiener diversity values ranged from 1.9-2.9 for benthic assemblages (Appendix 2.A) and 0.6-1.2 for drifting assemblages (Appendix 2.C). Lower diversity values in the drift appear to be due to lower taxonomic richness and higher proportions of dominant taxa present in drifting assemblages. Analysis of covariance detected higher benthic diversity values for the wet ecoregion ($p = 0.019$) (Appendix 2.B); no differences in diversity were detected for drift assemblages among any treatment categories.

2.5 Discussion

The presence of significant relationships between the benthos and drift depended upon the metric used to make the comparison. No relationships between total benthic and drifting densities existed among any treatment categories, which contrast with the findings of Siler *et al.* (2001) who reported a positive relationship between benthic abundance and drift density. They also noted that disparities in benthic abundance between their experimental treatments (exclusion of allochtonous inputs of riparian vegetation) and controls enhanced their ability to detect this trend. However, in this study no trend was detected despite differences in total benthic density among treatment categories.

Isolating EPT taxa and examining the total percent they represent of each assemblage revealed that a positive relationship between benthic and drift may not occur consistently throughout ecoregions. No relationship was detected in either logged or unlogged watersheds within the wet ecoregion despite the differences in the dry ecoregion. The lack of a relationship in the WU treatment may have been due, in part, to the high drift density of the caddisfly Limnephilidae, which increased the total % EPT present.

Community metrics and biotic indices that rely on presence-absence data, or tolerance values associated with invertebrates exposed to various environmental stressors, will likely differ between benthic and drifting assemblages because of the absence or scarcity of certain taxa in the drift. Hilsenhoff's family-level biotic index was selected because of its wide use as a measure of impairment to stream ecosystems using taxa specific tolerance values associated with environmental stressors, specifically organic enrichment

(Hilsenhoff, 1988). Although logging is not typically associated with enrichment, invertebrates that are intolerant of this stressor, such as the majority of stoneflies, are also intolerant of elevated temperatures and increased sedimentation associated with logging (Quinn *et al.*, 2004; Baillie *et al.*, 2005). More importantly, this analysis compared results of an index that categorized sites into discrete classes such as excellent, fair, and poor water quality. Of the 43% of sites misclassified by drift samples, 86% were classified into a higher water quality class than indicated by benthic samples. This appears to be due to the absence of tolerant invertebrates in the drift, particularly Diptera such as Muscidae, Empididae, and Dolichopodidae, common in benthic communities. This misclassification is evident in the downward shift in tolerance scores (Fig. 2.4a & b) and illustrates that drift may be taxa specific (Benke *et al.*, 1991). This point is further supported by the lower number of families present in the drift compared to the benthos. It is important to point out that not all of the families missing from that drift are considered tolerant. For instance, the stonefly Pteronarcyidae and caddisfly Glossosomatidae are highly intolerant of environmental stressors (Hilsenhoff, 1988) and were both rare in the drift. Lower numbers of families in the drift may also be reflected in the lower diversity scores of this assemblage because a substantial source of error in the Shannon diversity index and from a failure to include all taxa from the sampled community. This error increases as the proportion of taxa represented in the sample declines (Magurran, 1988).

Drift propensity can be measured at the level of individual taxa, taxonomic groups such as functional feeding groups, or community wide. The absence of a linear

relationship between the benthos and the drift suggests that drift rates may have been highly variable regardless of the benthic density. It may have been this high degree of variability that made the detection of any differences in community wide drift propensity difficult to see among treatment categories if such differences existed. While no differences were seen among treatment categories, the negative relationship between community wide drift propensity and benthic densities did support the findings of Siler *et al.* (2001) who stated that elevated drift in streams with lower benthic densities suggests a reduced carrying capacity forcing mobile individuals locate more abundant food resources. These results also support the findings of Lugthart & Wallace (1992) who stated that sub-lethal effects of stress (e.g., inadequate food or habitat resources) may increase invertebrate drift and decrease standing stocks.

This study has demonstrated that significant differences detected in composite metrics of benthic communities across ecoregions and logging histories were not seen in drifting assemblages from those same streams. Overall, differences in the mean values of assemblage metrics among treatment categories were detected for six of eleven metrics of benthic assemblages and one of eleven metrics of drift assemblages. In contrast, cluster analysis of benthic and drifting composition suggests that assemblage composition may be a useful tool for detecting differences in invertebrate community structure that composite metrics fail to reveal. The lower taxonomic richness of drift samples may have contributed to inability to detect differences among treatment categories. This reduced richness may have decreased the likelihood of seeing differences in community structure because fewer food acquisition behaviors and habitat requirements were

represented by the drift assemblage. All families of drifting scrapers identified in the dry ecoregion were also seen in the wet ecoregion with Heptageniidae being the dominant family in both. Drift density of all scraping taxa was similar between ecoregions with the exception of Heptageniidae, which was twice as abundant in the dry ecoregion and primarily responsible for the significant difference detected. This difference in drifting scrapers was not a result of differences in benthic scraper densities, suggesting that the drift propensity of this family may be higher in the dry ecoregion.

2.6 Conclusions

The results of this study suggest that drift assemblages are inconsistent in reflecting the densities or composition of benthic communities. This inconsistency occurred between ecoregions where metrics such as % EPT demonstrated a positive and significant relationship between benthic and drift assemblages within the dry ecoregion only. Differences in benthic densities and composition across ecoregions and logging histories were also not reflected in drift assemblages. Lower taxonomic richness in the drift appears to be largely responsible for the absence of relationships between it and the benthic community. For those taxa present in both assemblages, differences in the drift rates of individual taxa may be responsible for the absence of significant relationships between benthic densities and drift densities. As a result, it is recommended that drift samples not be used as a replacement for benthic samples in assessing the biological condition of invertebrate communities in headwater streams. However, drift sampling could be a useful tool if used in conjunction with benthic sampling to calculate drift

propensity, which has been shown to identify stream systems with potentially reduced food resources.

2.7 References

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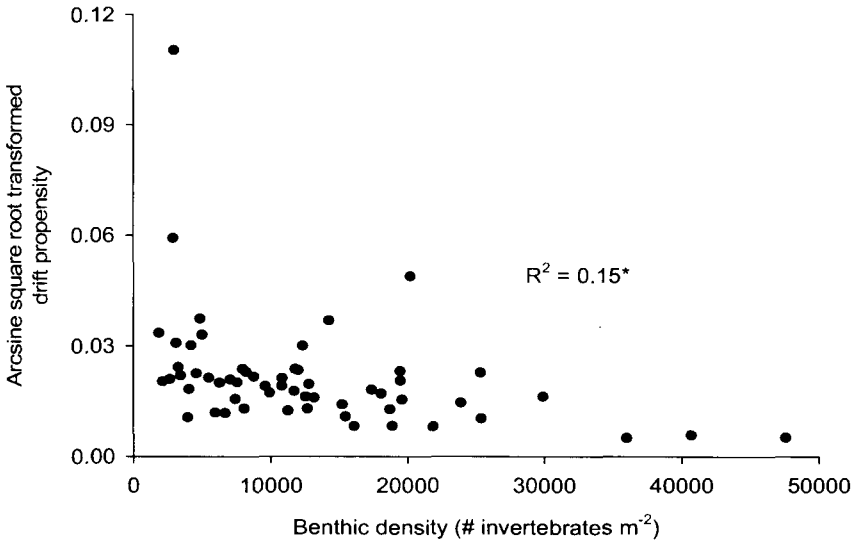

Fig. 2.1 Relationship between total benthic density and drift propensity (arcsine square root transformation) per site. * indicates a significant relationship.

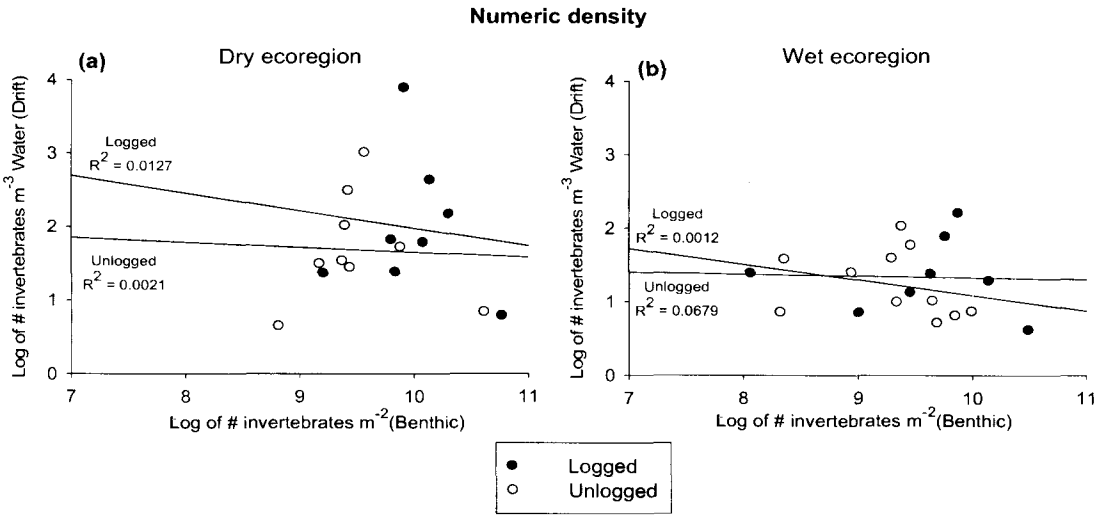

Fig. 2.2 Relationships between densities of benthic and drift assemblages in logged and unlogged watersheds of the (a) dry and (b) wet ecoregions. All relationships were non-significant at $\alpha = 0.05$.

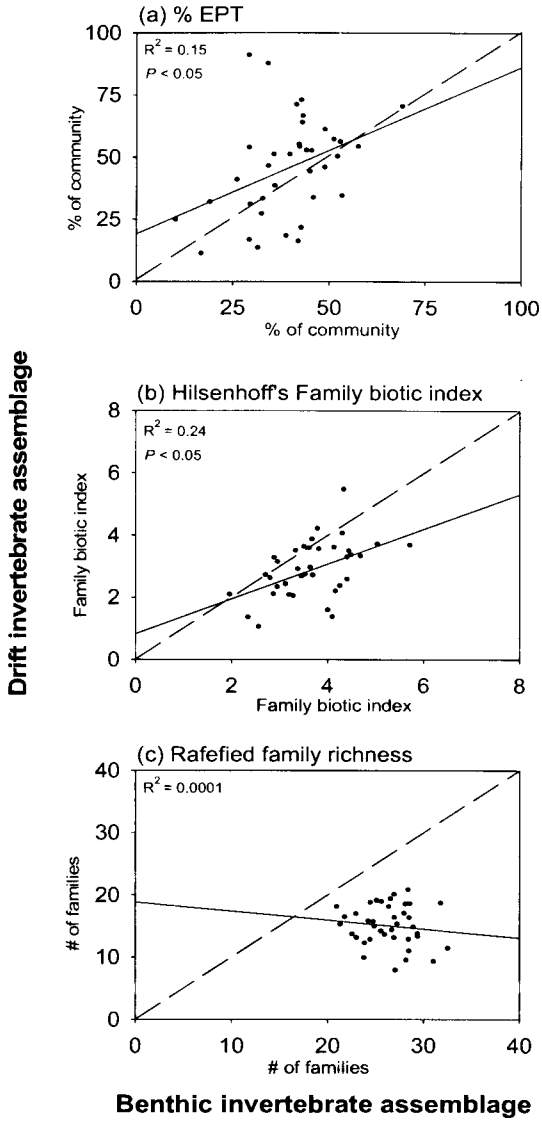

Fig. 2.3 Relationships between drift and benthos for selected invertebrate community measures. Solid lines indicate fitted regression and broken lines indicate a hypothetical 1:1 relationship between the benthos and drift.

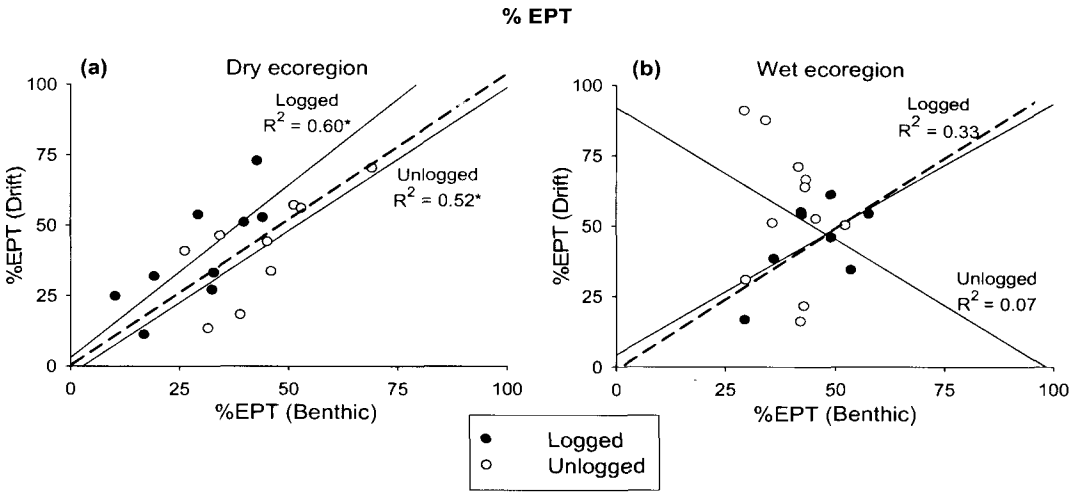

Fig. 2.4 Percent Ephemeroptera, Plecoptera, Trichoptera (EPT) taxa for benthic and drift assemblages in logged and unlogged watersheds of the (a) dry and (b) wet ecoregions. Solid lines indicate fitted regressions and broken lines indicate a hypothetical 1:1 relationship between the benthos and drift. * indicates a significant relationship at α = 0.05.

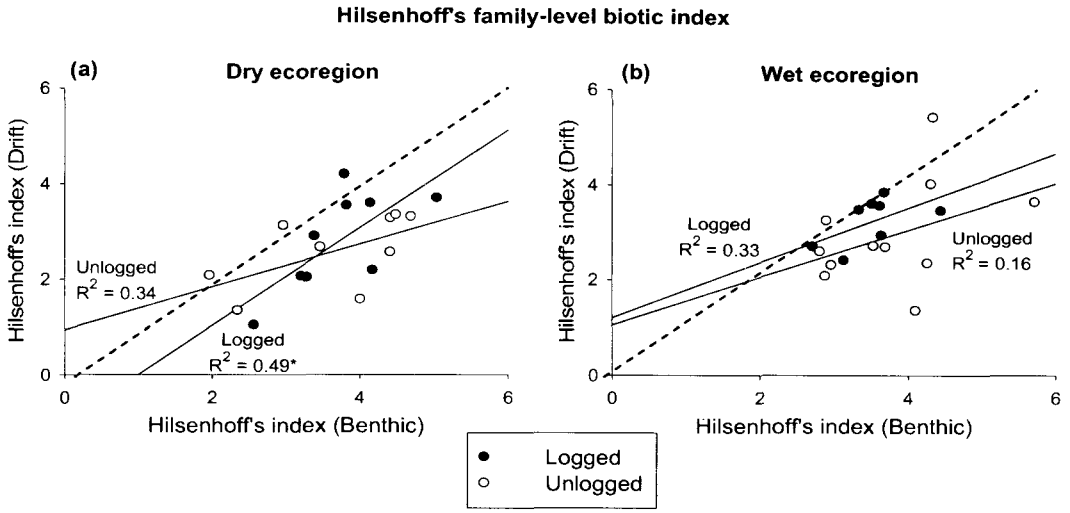

Fig. 2.5 Hilsenhoff's biotic family-level biotic index values for benthic and drift assemblages in logged and unlogged watersheds of the (a) dry and (b) the wet ecoregions. Solid lines indicate fitted regressions and broken lines indicate a hypothetical 1:1 relationship between the benthos and drift. * indicates a significant relationship at α = 0.05.

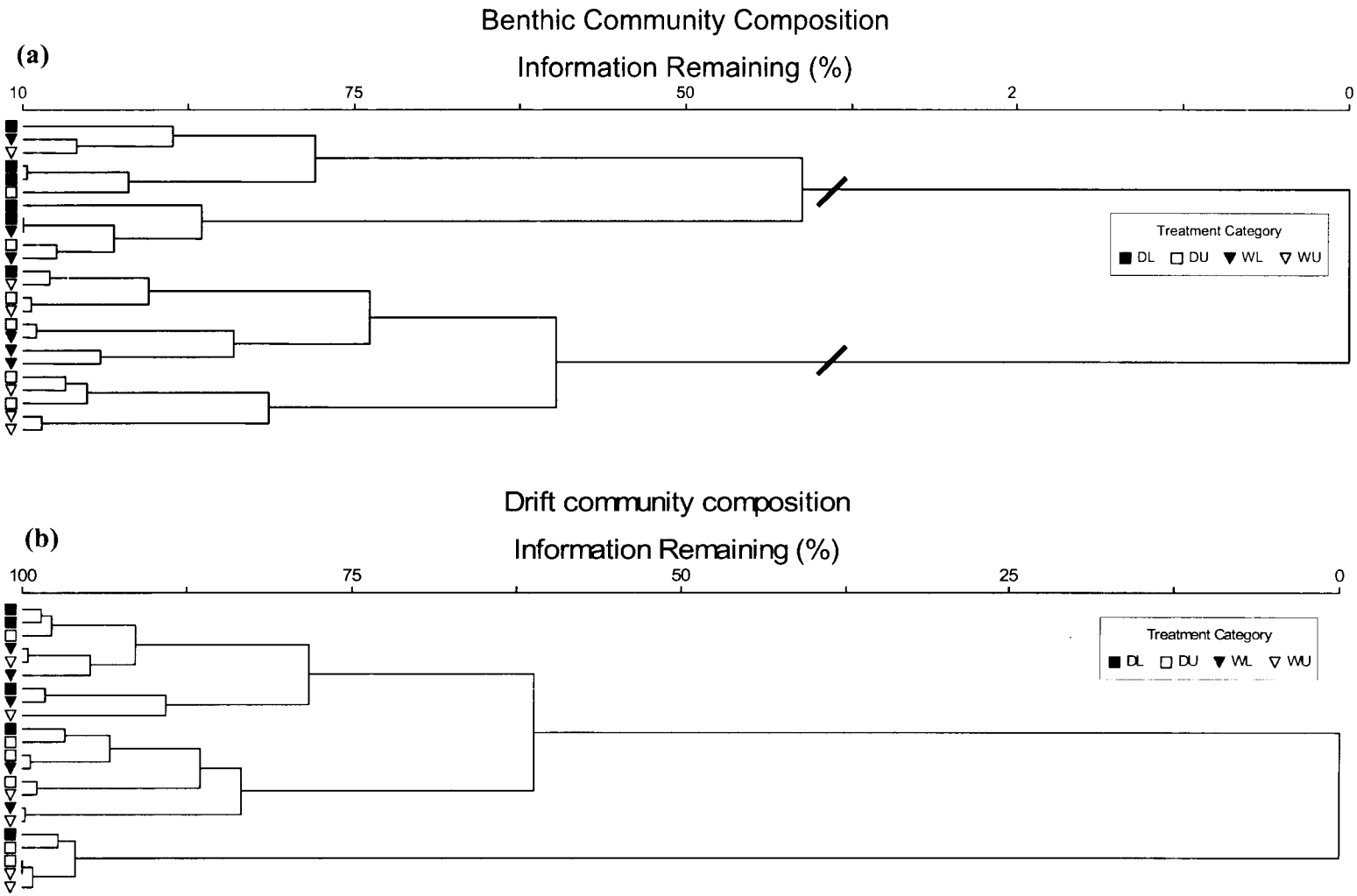

Fig. 2.6 Dendrogram from cluster analysis of total (a) benthic and (b) drift assemblage composition. Flexible beta method of clustering ($\beta = -1$) and a Sorensen (Bray-Curtis) distance measure were used. / symbol represents the location where the dendrogram was trimmed for analysis and similarities detected in community composition.

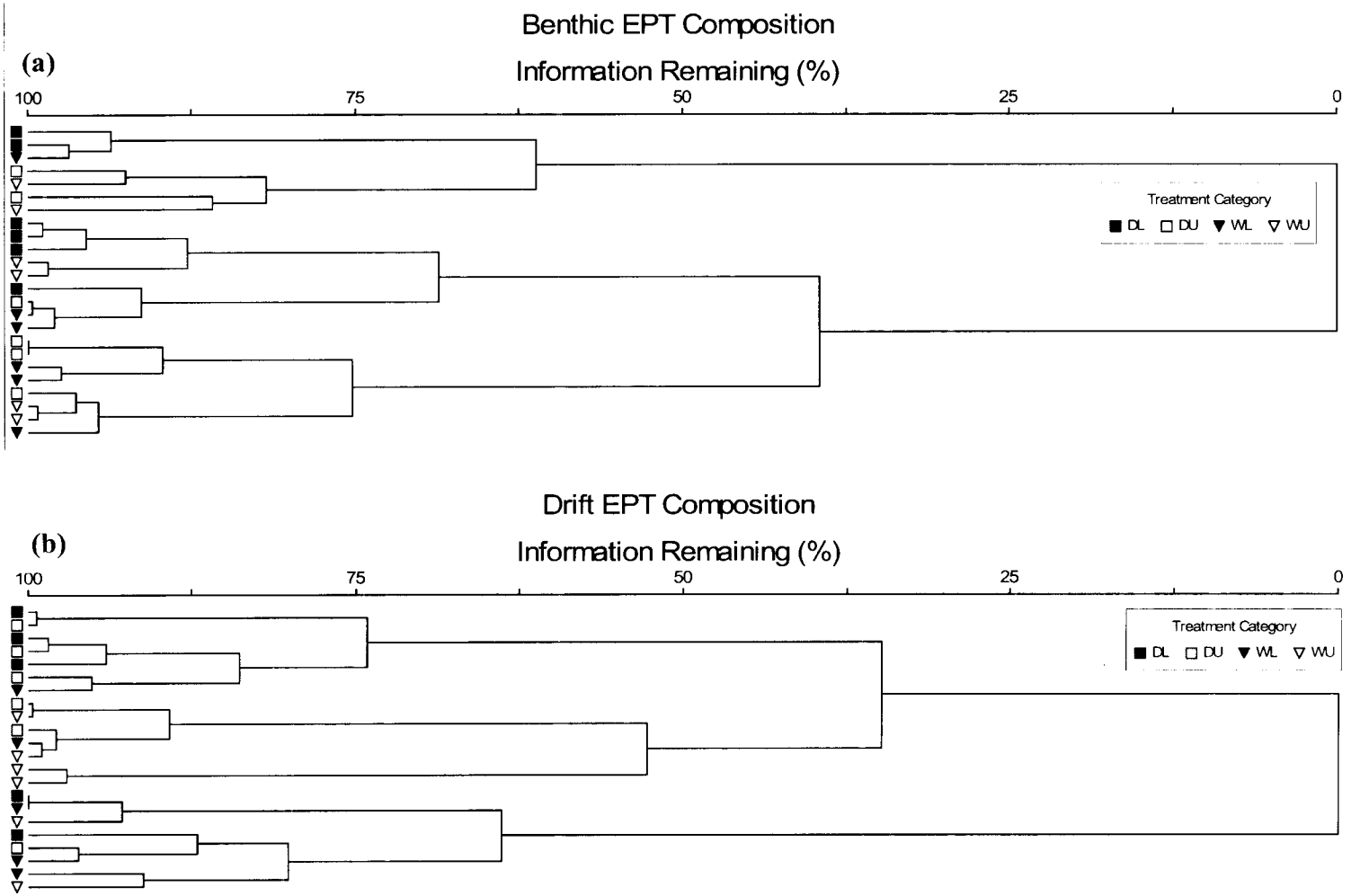

Fig. 2.7 Dendrogram from cluster analysis of EPT composition for (a) benthic and (b) drift assemblages. Flexible beta method of clustering ($\beta = -1$) and a Sorensen (Bray-Curtis) distance measure were used.

Table 2.1 Mean values for % dominant taxa, % EPT, rarefied family richness (families / category), diversity (Shannon-Wiener index), family biotic index (Hilsenhoff's biotic index), numeric density (benthic: individuals m^{-2}; drift: individuals m^{-3} water), for benthic and drifting invertebrates in four treatment categories. Standard errors in parentheses.

	Benthic assemblage				Drift assemblage			
	Dry-unlogged	Dry-logged	Wet-unlogged	Wet-logged	Dry-unlogged	Dry-logged	Wet-unlogged	Wet-logged
% dominant taxa	27 (2)	32 (4)	26 (3)	22 (2)	30 (4)	38 (6)	36 (5)	28 (4)
% EPT	43 (4)	31 (4)	40 (2)	43 (3)	42 (5)	39 (6)	52 (7)	46 (5)
Rarefied family richness	27 (1)	27 (1)	26 (1)	27 (1)	16 (1)	15 (1)	14 (1)	14 (1)
Shannon-Wiener diversity	2.59 (0.03)	2.41 (0.11)	2.65 (0.07)	2.63 (0.06)	1.02 (0.04)	0.93 (0.08)	0.95 (0.06)	1.00 (0.04)
Family biotic index	3.65 (0.34)	3.72 (0.24)	3.78 (0.27)	3.52 (0.18)	2.61 (0.21)	2.62 (0.36)	2.84 (0.33)	3.08 (0.23)
Numeric density	15705 (3001)	24772 (3526)	12342 (1787)	18298 (2940)	5.38 (1.83)	8.55 (4.53)	2.78 (0.57)	2.64 (0.80)

Table 2.2 Mean benthic density (individuals m^{-2}) and drift density (individuals m^{-3} water) for functional feeding groups within four treatment categories. Standard errors of values in parentheses.

	Benthic assemblage				Drift assemblage			
Functional feeding group	Dry-unlogged	Dry-logged	Wet-unlogged	Wet-logged	Dry-unlogged	Dry-logged	Wet-unlogged	Wet-logged
Filtering-collectors	134	56	216	184	0.05	0.04	0.07	0.05
	(61)	(20)	(70)	(46)	(0.02)	(0.02)	(0.03)	(0.02)
Gathering-collectors	4216	7885	3305	4852	2.43	4.49	0.72	1.56
	(677)	(1190)	(487)	(817)	(1.15)	(2.66)	(0.23)	(0.61)
Predators	740	837	713	763	0.50	0.74	0.26	0.55
	(140)	(135)	(152)	(117)	(0.24)	(0.28)	(0.10)	(0.31)
Scrapers	255	399	296	300	0.14	0.19	0.05	0.15
	(53)	(133)	(73)	(78)	(0.05)	(0.04)	(0.01)	(0.10)
Shredders	1316	1354	713	1685	0.53	0.58	0.54	0.58
	(404)	(264)	(149)	(307)	(0.31)	(0.27)	(0.27)	(0.31)

2.8 Appendices

Appendix 2.A

Mean assemblage metrics for benthic invertebrates for individual sites

Site name	Treatment category	Numeric density (no. m^{-2})	% Dominant taxa	% EPT	Rarefied family richness	Shannon-Weiner diversity	Hilsenhoff's biotic index
Hansel 6	dry-logged	24683	46	15	24	1.9	4.6
Hansel Rock	dry-logged	20241	42	17	30	2.2	2.6
Scotty	dry-logged	38802	38	31	31	2.3	4.0
South Shaser	dry-logged	17270	23	41	30	2.8	3.3
Trib Mid Shaser	dry-logged	37802	16	39	32	2.8	2.9
Tronsen East	dry-logged	14082	26	39	30	2.5	3.6
Allen	dry-unlogged	15742	32	43	31	2.6	4.5
Hansel 1	dry-unlogged	12412	30	39	30	2.7	4.0
King	dry-unlogged	23754	21	57	24	2.7	3.9
Mission Top	dry-unlogged	12639	28	26	31	2.5	4.7
Sand 1	dry-unlogged	13299	27	41	28	2.6	3.7
Tronsen 2	dry-unlogged	13204	24	39	24	2.5	2.2
Dirty Face	Wet-logged	7989	21	47	27	2.5	4.0
Fir	Wet-logged	12808	32	42	33	2.7	3.6
Gill	Wet-logged	30766	16	51	28	2.8	2.9
Hidden	Wet-logged	15298	16	49	31	3.0	3.7
Roaring b	Wet-logged	19543	19	30	24	2.5	3.3
Upper Trout Trib	Wet-logged	22504	22	36	28	2.4	3.9
Butcher East	Wet-unlogged	14034	41	36	27	2.4	5.0
End Ice 1	Wet-unlogged	11554	20	39	26	2.8	3.9
End Ice 3	Wet-unlogged	21957	23	29	29	2.9	4.3
Horse	Wet-unlogged	7819	16	45	25	2.8	2.9
Sears	Wet-unlogged	13223	27	47	30	2.7	3.6
SLW	Wet-unlogged	10271	30	39	24	2.5	3.2

Appendix 2.B

Results for repeated measures ANCOVA for assemblage metrics of benthic and drifting invertebrates

Invertebrate density for functional feeding groups	Benthic assemblage (no. m^{-2}) F ratio			Drift assemblage (no. m^{-3} water) F ratio		
	Logging	Ecoregion	Logging x Ecoregion	Logging	Ecoregion	Logging x Ecoregion
Filtering-collectors	0.09	9.18**	1.42	0.00	1.05	1.40
Gathering-collectors	4.93*	1.87	0.63	1.17	0.66	0.01
Predators	0.41	0.00	0.02	2.01	0.64	0.33
Scrapers	1.13	0.31	0.42	1.36	7.75*	0.05
Shredders	4.51*	0.01	5.21*	2.54	1.15	0.76
Assemblage metric						
% dominant taxa	0.53	5.64*	3.76	0.10	0.01	2.21
% EPT	0.77	2.59	5.73*	0.40	1.24	0.48
Rarefied family richness	1.04	1.55	0.34	0.18	0.88	0.32
Diversity	1.66	5.27*	1.55	0.01	0.04	1.46
Hilsenhoff's biotic index	0.09	0.45	0.23	0.97	0.36	0.03
Numeric density	5.81*	0.07	0.01	1.53	2.14	0.13

Note: * = $P < 0.05$, ** = $P < 0.01$

Appendix 2.C

Mean assemblage metrics for drifting invertebrates for individual sites

Site name	Treatment category	Numeric density (no. m^{-3} water)	% Dominant taxa	% EPT	Rarefied family richness	Shannon-Weiner diversity	Hilsenhoff's biotic index
Hansel 6	dry-logged	6.8	29	28	17	1.0	3.6
Hansel Rock	dry-logged	32.9	67	23	9	0.6	1.0
Scotty	dry-logged	3.6	38	40	15	0.9	3.2
South Shaser	dry-logged	3.0	23	62	17	1.1	2.0
Trib Mid Shaser	dry-logged	na	na	na	na	na	na
Tronsen East	dry-logged	3.5	32	43	16	1.0	3.2
Allen	dry-unlogged	4.1	18	52	19	1.2	2.9
Hansel 1	dry-unlogged	5.4	42	24	13	0.9	2.1
King	dry-unlogged	1.3	24	57	17	1.1	3.0
Mission Top	dry-unlogged	5.9	23	50	16	1.1	3.1
Sand 1	dry-unlogged	2.8	30	56	18	1.1	3.1
Tronsen 2	dry-unlogged	9.0	42	23	14	0.9	1.7
Dirty Face	Wet-logged	1.8	30	46	13	1.0	3.5
Fir	Wet-logged	5.7	17	54	17	1.1	3.3
Gill	Wet-logged	2.8	22	48	15	1.1	2.6
Hidden	Wet-logged	3.1	39	46	14	0.9	3.8
Roaring b	Wet-logged	8.2	40	35	13	0.9	2.6
Upper Trout Trib	Wet-logged	na	na	na	na	na	na
Butcher East	Wet-unlogged	4.8	36	26	12	0.9	4.6
End Ice 1	Wet-unlogged	1.1	39	33	15	0.9	2.0
End Ice 3	Wet-unlogged	2.0	35	56	13	0.9	2.0
Horse	Wet-unlogged	3.3	20	59	18	1.2	2.2
Sears	Wet-unlogged	2.5	36	60	16	1.0	3.3
SLW	Wet-unlogged	2.9	49	75	12	0.8	3.0

Appendix 2.D

Mean functional feeding group density (individuals m^{-3} water) for drifting assemblages of individual sites

Site name	Treatment	Filtering-collectors	Gathering-collectors	Predators	Scrapers	Shredders
Hansel 6	dry-logged	0.02	3.80	1.79	0.14	1.45
Hansel Rock	dry-logged	0.11	18.09	0.94	0.27	1.26
Scotty	dry-logged	0.04	2.98	0.43	0.16	0.18
South Shaser	dry-logged	0.00	0.47	0.16	0.26	0.44
Trib Mid Shaser	dry-logged	na	na	na	na	na
Tronsen East	dry-logged	0.07	1.43	0.59	0.21	0.72
Allen	dry-unlogged	0.03	0.38	0.20	0.10	0.17
Hansel 1	dry-unlogged	0.02	3.65	0.53	0.21	0.33
King	dry-unlogged	0.03	0.30	0.09	0.08	0.26
Mission Top	dry-unlogged	0.02	0.64	0.43	0.09	0.87
Sand 1	dry-unlogged	0.10	1.13	0.21	0.14	0.17
Tronsen 2	dry-unlogged	0.09	7.20	1.63	0.27	1.80
Dirty Face	Wet-logged	0.05	1.16	0.33	0.08	0.36
Fir	Wet-logged	0.12	3.25	1.77	0.56	1.84
Gill	Wet-logged	0.06	0.72	0.33	0.02	0.24
Hidden	Wet-logged	0.04	1.65	0.32	0.18	0.75
Roaring b	Wet-logged	0.05	4.58	0.69	0.05	1.22
Upper Trout Trib	Wet-logged	na	na	na	na	na
Butcher East	Wet-unlogged	0.03	1.07	0.62	0.05	0.27
End Ice 1	Wet-unlogged	0.02	0.65	0.14	0.08	0.12
End Ice 3	Wet-unlogged	0.06	0.60	0.09	0.04	0.15
Horse	Wet-unlogged	0.02	0.48	0.13	0.07	0.18
Sears	Wet-unlogged	0.21	1.29	0.21	0.05	0.25
SLW	Wet-unlogged	0.06	0.47	0.32	0.01	2.07

Appendix 2.E

Mean functional feeding group density (individuals m^{-2}) for the benthic assemblages of individual sites

Site name	Treatment	Filtering-collectors	Gathering-collectors	Predators	Scrapers	Shredders
Hansel 6	dry-logged	0	8816	600	68	984
Hansel Rock	dry-logged	28	7272	552	176	564
Scotty	dry-logged	104	12620	1336	444	1996
South Shaser	dry-logged	32	4604	772	956	988
Trib Mid Shaser	dry-logged	192	11008	1312	736	2784
Tronsen East	dry-logged	36	4248	548	72	1132
Allen	dry-unlogged	80	4420	672	176	1332
Hansel 1	dry-unlogged	96	3552	616	344	656
King	dry-unlogged	356	5594	1076	406	2644
Mission Top	dry-unlogged	120	4016	680	136	416
Sand 1	dry-unlogged	92	3612	844	240	852
Tronsen 2	dry-unlogged	36	3672	460	216	1220
Dirty Face	Wet-logged	254	2088	184	156	746
Fir	Wet-logged	220	3148	668	164	1232
Gill	Wet-logged	280	8216	1144	712	2752
Hidden	Wet-logged	160	3664	784	312	1568
Roaring b	Wet-logged	0	6176	896	32	1184
Upper Trout Trib	Wet-logged	88	5888	980	300	2320
Butcher East	Wet-unlogged	324	4120	412	480	636
End Ice 1	Wet-unlogged	388	3012	772	174	556
End Ice 3	Wet-unlogged	192	6096	1520	784	720
Horse	Wet-unlogged	150	2032	450	296	400
Sears	Wet-unlogged	164	3624	476	264	1104
SLW	Wet-unlogged	66	2346	1056	24	866

General Conclusions

The findings of this study suggest that for benthic communities the numeric abundance of individuals and taxonomic groups such as scrapers, shredders, gathering-collectors and EPT taxa differ between wet and dry ecoregions and between logged and unlogged watersheds. However, these differences in benthic invertebrates were not consistently reflected in drift assemblages; thus, drift samples may not serve as an adequate substitute for benthic samples in assessing the biological condition of headwater streams.

Composite metrics such as total numeric abundance, % EPT, and diversity revealed differences in benthic assemblages but not in drift assemblages. Taxonomic composition on the other hand was similar among sites within ecoregions for both assemblages, and further analysis of these compositional similarities may help to identify whether individual taxa are primarily responsible for these patterns. Sample size and taxonomic resolution appear to be important in identifying similarities in community composition of invertebrate assemblages among sites and treatment categories. Chapter 1 utilized five months of data with all taxonomic identification conducted to the genus level. In contrast, Chapter 2 utilized two months of data and invertebrates were identified to family. These differences in sample size and taxonomic resolution were likely responsible for differing patterns in EPT composition of benthic samples.

This study has demonstrated that the outcomes of biological assessments of the condition of headwater stream catchments exposed to timber harvest may be driven by climate and vegetation cover (i.e., ecoregion). It has also demonstrated that benthic

samples are a more sensitive measure of change in invertebrate communities than are drift samples, and thus, drift sampling alone should not be used in drawing conclusions about headwater stream condition. Nonetheless, resource managers might consider using regional classifications and invertebrate metrics such as functional feeding group, which appear to respond to differences in riparian forest composition, as a potential tool in detecting changes to aquatic communities following commercial timber harvest.